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Oosterhuis, F.H.; Brouwer, R.

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**Benchmark development for the proportionality  
assessment of PBT and vPvB substances**

Frans Oosterhuis

Roy Brouwer

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Rijksinstituut voor Volksgezondheid  
en Milieu  
*Ministerie van Volksgezondheid,  
Welzijn en Sport*



IVM Institute for  
Environmental Studies



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**IVM**

Institute for Environmental Studies  
VU University Amsterdam  
De Boelelaan 1087  
1081 HV AMSTERDAM  
The Netherlands  
T +31-20-598 9555  
F +31-20-598 9553  
E [info.ivm@vu.nl](mailto:info.ivm@vu.nl)

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## Executive summary

This report is intended to provide the Committee for Socio-Economic Analysis (SEAC) of the European Chemicals Agency (ECHA) with information that could be used in the development of a benchmark for assessing the proportionality of restriction proposals and authorization applications for persistent, bioaccumulative and toxic (PBT) as well as very persistent, very bioaccumulative (vPvB) substances (henceforth: PBTs). For these substances a full cost-benefit assessment is usually not feasible due to their specific properties. Instead, a SEAC working group has proposed to use a cost-effectiveness framework, which implies the need for a benchmark.

To support the development of a benchmark, information was gathered on the costs of reducing stocks (presence in 'technosphere' or environment) and flows (emissions) of PBTs. This cost information is then related to decision making on PBT reduction. If a certain proposed measure has been rejected due to excessive costs per unit of reduction, this can be seen as an indication for society's 'maximum willingness to pay'. The exceedance of this cost level might then be considered to be disproportional in comparable future cases as well. Clearly, this approach rests on a number of assumptions and neglects any differences in properties between PBTs. It should therefore be seen as no more than a first step towards benchmarking.

The following substances were selected as cases for the present study: D4/D5, decaBDE, HBCDD, HCB, HCH, PCBs, PFOA and PFOS. An analysis of existing documentation containing estimates of the cost-effectiveness of measures to reduce the stocks or flows of these PBTs led to the following observations:

- the range of cost estimates covers several orders of magnitude, from less than EUR 1 per kg (sometimes even negative costs) to several millions of euros per kg;
- a wide variety of cost estimates is also visible within each of the three types of measures distinguished (substitution, emission control and clean-up/remediation), but the lowest estimates are found among the 'substitution' measures;
- costs per kg avoided emission are substantially higher than per kg avoided substance use;
- the cost types that are included in the estimates differ widely: sometimes it is just the price difference (with a 'drop-in' substitute), whereas in other cases for instance the costs of (additional) R&D, capital loss, adaptation of equipment and/or indirect costs such as market loss are included;
- specific cost items such as quality loss (in case of substituting with an inferior substance) or reduced social benefits from recycling (in case of a ban without exemptions) are included only in a few studies;
- the costs per kg tend to be relatively low if the total amount or the concentration of the PBT is high, and if it is used or stored in a closed/confined space, whereas the costs per kg can become very high if the PBT is dilute or widely dispersed.

Decisions on PBT measures appear to be rarely explicitly motivated by cost effectiveness arguments. In particular, it is hard to find clear statements that a restriction or ban on the use of a PBT or the clean-up of a site polluted by a PBT should be abandoned due to disproportionate or excessive costs per kg. Nevertheless, the available evidence suggests that there is a wide 'grey zone' (orders of magnitude between EUR 1000 and EUR 50,000 per kg avoided PBT use/presence or emission) within which the cost of a measure can either be 'acceptable' or 'too high'. Within this

‘grey zone’ the outcome of the decision making may depend on substance-specific and situation-specific conditions and on other considerations than cost-effectiveness (e.g. affordability and competitiveness).

As a conclusion, this study has led to a substantial evidence base in terms of cost effectiveness data, but the role that these data play or have played in decision making is much less clear, as explicit statements on ‘disproportional costs’ are scarce. There seems to be still quite a way to go until the envisaged benchmarking. With a view to further work in this area, it is recommended to:

- ensure that cost estimates for PBT related measures are complete (all relevant cost types included);
- specify the assumptions made in estimating the costs;
- indicate the nature of the estimate (in EUR per kg of use/presence reduction or in EUR per kg of emission reduction);
- specify the additional considerations that may play a role in the proportionality assessment (besides cost effectiveness);
- further expand the database that was prepared within the framework of the present study, and keeping it up-to-date;
- investigate whether the envisaged benchmarking approach could possibly also be applicable to other (non-PBT) substance categories;
- embark on additional efforts to construct cost effectiveness curves showing the amount of each PBT that can be substituted/removed at a certain cost level.

## Abbreviations

BCD	base catalyzed decomposition
BFR	brominated flame retardant
CEPA	Canadian Environmental Protection Act
D4	octamethylcyclotetrasiloxane
D5	decamethylcyclopentasiloxane
d(eca)-BDE	decabromodiphenylether
EBP	ethane,1,2-bis-pentabromophenyl
ECHA	European Chemicals Agency
EEE	electrical and electronic equipment
EPA	Environmental Protection Agency (USA)
EPS	expanded polystyrene
FR	flame retardant
HBCDD	hexabromocyclododecane
HCB	hexachlorobenzene
HCH	hexachlorocyclohexane
HIPS	high impact polystyrene
IVM	Instituut voor Milieuvraagstukken (Institute for Environmental Studies), VU University Amsterdam
PBDE	polybromodiphenylether
PBT	Persistent, bioaccumulative and toxic
PCBs	polychlorinated biphenyls
PFOA	perfluorooctanoic acid
PFOS	perfluorooctane sulfonate
pFR	polymeric flame retardant
PIR	polyisocyanurate
POP	persistent organic pollutant
PPE	polyphenylene ether
ppm	parts per million
PUR	polyurethane
REACH	Registration, Evaluation and Authorisation of CHemical substances
RIVM	Rijksinstituut voor Volksgezondheid en Milieu (National Institute for Public Health and the Environment, The Netherlands)
RoHS	Restriction of Hazardous Substances
SEAC	Socio-Economic Analysis Committee



SVHC	Substance(s) of Very High Concern
vPvB	very persistent, very bioaccumulative
XPS	extruded polystyrene

## 1 Introduction

This report was prepared within the framework of a project commissioned by the Dutch Ministry of Infrastructure and the Environment. It is intended to provide the Committee for Socio-Economic Analysis (SEAC) of the European Chemicals Agency (ECHA) with information that could be used in the development of a possible benchmark for assessing the proportionality of restriction proposals and authorization applications for PBT and vPvB substances.

### 1.1 Background of the study

Persistent, bioaccumulative and toxic (PBT) as well as very persistent, very bioaccumulative (vPvB) substances are two categories of substances of very high concern (SVHC) under the EU's chemicals legislation (REACH). As with other SVHCs, restriction proposals and authorization applications for PBT and vPvB substances<sup>1</sup> have to be assessed with respect to their proportionality. However, for these substances a full cost-benefit assessment is usually not feasible. Persistent and bioaccumulative properties allow substances to accumulate in remote environments, which is a process difficult to reverse. Cessation of emissions will not immediately result in a reduction in chemical concentration levels due to the long half-life (Verhoeven *et al.*, 2012). As a result, the potential impacts of PBT and vPvB substances can occur far away from their source, both in space (they tend to spread in the environment, sometimes over long distances) and time (due to their persistency). Therefore, these impacts cannot be predicted accurately and hence it is more often than not impossible to quantify health and environmental impacts caused by the emission of and exposure to these substances. In principle, a 'safe' concentration level for PBT substances in the environment cannot be established with sufficient reliability (Verhoeven *et al.*, 2012).

A SEAC working group is developing an evaluation framework for the proportionality assessment of restriction proposals and authorization applications of PBT substances. The proposal of the working group is to use a cost-effectiveness framework, rather than cost benefit analysis, for SEAC's evaluation of PBT and vPvB cases (SEAC, 2014). In a cost-effectiveness analysis, the cost of a certain (policy) measure is related to some non-monetary parameter, for instance the amount of emission reduction that can be achieved with this measure. To be able to conclude on the proportionality of policy measures based upon a cost-effectiveness assessment, there is a need for a benchmark: the decision maker (or in this case SEAC as a scientific advisory committee) wants to know if a specific level of cost (e.g. per unit of emission reduction) should be considered as proportional or disproportional.

One possible approach of developing such a benchmark is by gathering information on the level of cost that society is spending (or has spent in the past) on PBT emission reduction (or on reductions in the use of, or the exposure to these substances). The basic idea behind this 'revealed preference' approach is that this information gives indications of the 'public willingness to pay' for such reductions. If a certain proposed measure has been rejected due to excessive costs per unit of reduction, this can be seen as an indication for society's 'maximum willingness to pay', and hence as a benchmark, the exceedance of which would be considered disproportional, possibly also in comparable future cases.

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<sup>1</sup> In the remainder of this report, we will use the term 'PBT (substances)' for both categories together.

Clearly, this approach rests on a number of assumptions, including the idea that policy making is a rational process, guided by consistent considerations and focusing on cost-effectiveness only (ignoring other societal objectives such as distributional and/or equity issues). Moreover, applying the approach to PBT substances in general possibly neglects the differences in properties within this group of substances as well as differences in the specific circumstances of a restriction proposal or authorization application. Given furthermore the large uncertainties surrounding the cause-effect chain, this may imply that a set or a range of benchmarks rather than a single benchmark would be appropriate for the proportionality assessment of PBT substances. As a first possible step towards such benchmarking, available evidence on the cost effectiveness of measures addressing a range of these substances is collected. In a second step this information is analysed and assessed on its relevance to proceed towards circumstantial/conditional benchmarking.

## 1.2 Objective

The present study aims at setting some initial steps towards benchmarking the proportionality assessment of PBT restriction proposals and authorization applications by gathering information on the cost-effectiveness of (policy) measures that have been proposed and accepted or rejected to restrict the use and emissions of and exposure to these substances. The cost-effectiveness figures are accompanied by specific information on the circumstances and conditions prevailing in each case, in order to assess the general applicability of such a set of benchmarks related to particular situations.

## 1.3 Approach, methodology and limitations

The approach taken in this study is based on the assumption that measures to avoid the use of PBTs, their emissions or their presence in products and in the environment are taken by governments taking into consideration the cost effectiveness of these measures. If one furthermore assumes that the decision making on such measures is 'rational' in the sense that measures with a low cost per kg (use or emission reduction) are preferred over those with a higher cost, then the available information on cost effectiveness and decisions taken could be used as indicators for the amount society is prepared to pay for such reductions. The highest costs per kg actually paid (or the lowest cost level per kg which have in actual decision making been considered as too high) would then point to the order of magnitude where the cost of PBT reduction would still be proportionate (respectively become disproportionate).

Of course, this is a simplification of reality in several respects. Public decision making is based on many other considerations than just cost effectiveness, and sometimes reliable information on cost effectiveness is not available at all. Measures with relatively low costs per kg may not be carried out due to a lack of available budget or disagreement on the question who should pay the bill. Measures with very high costs per kg, on the other hand, may still be carried out because of, for instance, political pressure, or existing legal requirements.

Furthermore, in this approach PBTs are treated collectively as if they were a homogeneous group of substances, which they are obviously not. Each PBT has its own specific properties and applications, but for the purpose of this report (comparing cost effectiveness estimates for PBT reduction) these differences are left out of consideration. A caveat is therefore appropriate when interpreting the results.

A number of substances were selected as cases for the present study. Criteria for the selection were:

- PBT, vPvB and/or POP characteristics of the substance generally acknowledged;
- the substances should be subject to existing (or past) policies/measures (not necessarily under REACH);
- they should be representative for substances/substance groups that are currently relevant;
- a variety in application and emission/diffusion patterns (e.g.: closed/open; dispersed/non-dispersed; point/non-point);
- preferably the set should include one or more substances with just PBT/vPvB features (i.e. not be a SVHC for human toxicology reasons);
- data availability;
- EU focus.

This resulted in the following set: D4/D5, decaBDE, HBCDD, HCB, HCH, PCBs, PFOA and PFOS. For each of these substances, documentation was collected and scrutinized for information on the cost effectiveness of measures to control them. This included policy related documents (such as restriction reports and underlying studies) as well as journal articles and various internet based sources. The majority of sources was European (80%), but information from other continents (mainly North America) was used as well. In some cases, additional calculations were made to arrive at harmonized and comparable figures. The collected data were put in an Excel database, in which also a number of standardization calculations were done (e.g. adjusting different currencies from various years for inflation and converting everything in Euros in 2014 price levels).

## 1.4 Outline of the report

Chapter 2 discusses the different types of costs that can be distinguished along the life cycle of (products/processes with) PBTs. Chapter 3 summarizes the findings on cost effectiveness estimates for the selected substances and the main factors that can explain their variation. Chapter 4 addresses the role of the cost estimates in decision making on the selected PBTs. In chapter 5 we briefly discuss the possible relevance and implications of our findings for the development of proportionality benchmarks in the framework of REACH. Chapter 6 presents conclusions and some suggestions for further work.



## 2 Different cost types

When discussing the cost of reducing PBTs, it is useful to be aware that there are various ways of managing these substances, related to the stage in the life cycle of the product or process in which they perform a function or their presence in the environment (see Table 1). Broadly speaking, we can make a distinction between the development/design stage, the service life<sup>2</sup> (during which the PBT performs its function in the product or process), and the waste stage (which may be reached earlier than the end of the product's useful life if it is decided to prohibit or restrict the use of the substance). In the development/design stage, substitution (by a different substance, product or process) is the main approach for PBT control. During the service life of a product containing the PBT (or a process in which the PBT is used) controlling the substance will usually require the clean-up or decontamination of the products and installations (and replacement by a non-PBT substance<sup>3</sup>); in some cases emission reduction measures may be a (temporary) solution. In the waste stage, the PBT will have to be disposed of in a safe manner (for instance incineration or other (bio)chemical treatment; sometimes controlled disposal/landfilling). Furthermore, if the substance is present in the environment, for instance due to past emissions or uncontrolled disposal, techniques will be required to remove it from the soil or (ground)water before it can be disposed of.

Different types of costs are related to each of these stages and situations. The present report makes a basic distinction between three categories: the costs of substitution, of emission control, and of remediation, clean-up and disposal. The costs may not only encompass additional expenditure, but also reduced benefits or welfare, for example if the substitute has a lower quality or reduced performance compared to the original PBT. Furthermore, there may also be various types of indirect costs associated with PBT control, for instance due to the fact that a ban on the use of a substance (without exemptions) implies that a product that contains the substance cannot be recycled and has to be incinerated or disposed of otherwise.

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<sup>2</sup> PBTs may be contained in articles, product formulations or combinations thereof. For use in this report 'service life' is considered to apply to any of these applications.

<sup>3</sup> Obviously, this replacement can also be seen as a kind of substitution; the distinction is therefore not as clear-cut as it might seem. For pragmatic reasons, the cost of replacing a substance during the service life of a product or installation is included under 'cost of disposal, emission control and remediation' in this report.

Table 1 Different cost types for PBT control along the life cycle

	Development / design	Service life / use	Waste	Environment
Process to control or remove the PBT	Substitution (substance or product); process change	Emission reduction Clean up / decontamination	Safe disposal	Remediation from environmental compartment
Example 1 (HBCDD)	Producing new EPS with an alternative FR or replacement of EPS by an alternative insulation material	Removing installed EPS insulation panels that contain HBCDD from buildings	Incineration of materials with HBCDD	Sanitation of construction waste dumps
Example 2 (PCBs)	Using alternatives to PCBs in new transformers / capacitors	Removing PCBs from existing stock of transformers / capacitors	Incineration of PCB waste and soil/sludge with PCBs	Removing PCBs from contaminated sites
Example 3 (PFOS)	Introducing a PFOS-free process in electroplating	Replacing PFOS in existing fire fighting installations; minimization of emissions in applications that are still allowed	Thermal treatment or controlled landfilling of PFOS waste and soil/sludge with PFOS	Purification of surface water used for water supply; removing PFOS from soil and groundwater polluted by fire fighting foam
Associated direct costs	<ul style="list-style-type: none"> <li>• Additional development and production cost</li> <li>• Cost of quality / performance loss</li> </ul>	<ul style="list-style-type: none"> <li>• Clean up and replacement cost</li> <li>• Cost of emission control</li> </ul>	<ul style="list-style-type: none"> <li>• Cost of collection and separation</li> <li>• Cost of the various waste treatment routes</li> </ul>	Remediation cost
Possible indirect costs	Loss of competitiveness (turnover, employment) due to cost increases	Unavailability of equipment or infrastructure during decontamination operations	Foregone recycling opportunities	Unavailability of land or water until remediation is completed
Cost effectiveness	Costs can be expressed per kg avoided/reduced use or per kg avoided/reduced emission			

As the last row of Table 1 indicates, cost effectiveness estimates for PBT control can be expressed basically in two different ways: per kg avoided or reduced use (or reduced presence in products or the environment), and per kg avoided or reduced emission. The first relates to the total **stock** of a PBT (either in the ‘technosphere’ or in the environment) for instance the amount of PCBs that is present in a transformer or at a specific polluted site. The second relates to the **flow** of the PBT from the ‘technosphere’ to the environment, for instance the amount of PFOS emitted by an electroplating plant that can be prevented by switching to an alternative process or substance. The difference between the two may be significant, especially in cases where a prohibition is considered for a substance that is being used in ‘closed’ applications or with adequate emissions/leakage control measures: in such cases the cost per kg used may be relatively low, but the cost per kg emission avoided can be very high (due to the low initial level of emissions). In section 3.1 and 3.2 the two types of cost estimates are treated together, but in section 3.3 we analyze the difference between the two types and in the remainder of the report we will specify whether cost estimates relate to avoided/reduced use or avoided/reduced emission.

### 3 Cost effectiveness of PBT control measures

This chapter summarizes the findings on cost effectiveness estimates from the literature. Details per substance are presented in the Annex. The focus in this chapter is on the main findings regarding the levels of cost effectiveness estimates and the factors that can explain their variation.

#### 3.1 Cost estimates cover several orders of magnitude

A superficial look at the full set of collected cost-effectiveness estimates suggests that the cost of measures reducing the risk of PBT substances can range between less than 1 euro per kg (sometimes even negative costs, i.e. potential savings for example because a cheaper practice or technology is available which has not yet been fully adopted by all parties involved) and several millions of euros per kg. These wide ranges can even be observed for a single PBT substance (see Table 2).

*Table 2 Summary statistics: substitution, emission control and remediation costs (in EUR per kg)\**

Substance	Mean	Standard error	Median	Min-max	N
D4/5	75.6	44.3	27.5	-4 - 399	9
deca-BDE	29,687.5	18,433.0	326.9	1 - 251,281	14
HCB	54.2	51.3	3.8	1 - 208	4
HCH	159.6	123.0	12.5	1 - 760	6
HBCDD	1,290.0	868.1	24.9	-194 - 10,114	14
PCBs	45,801.0	40,883.1	907.5	1 - 413,200	10
PFOA	1,580.7	673.8	1,507.5	28 - 3,281	4
PFOS	1,213,571.5	937,814.5	7,702.5	1 - 21,412,950	23

\* Please note that there can be slight differences between the figures in the statistical analysis in this section and those in the remainder of the report. The former are taken from the database, where they have undergone standardization to allow for inflation and exchange rate changes. Figures mentioned elsewhere in the report are those from the original sources, converted with fixed (current) exchange rates.

The summary statistics in Table 2 are based on 36 studies generating 84 values from 10 individual countries, most of which (79.8%) originate from or cover Europe (Figure 1), over a time period of more than 25 years (1989-2015), as can be seen from Figure 2. Although there is a small peak between 2004 and 2006 (25% of the observations), most studies (60.0%) were carried out after 2009.



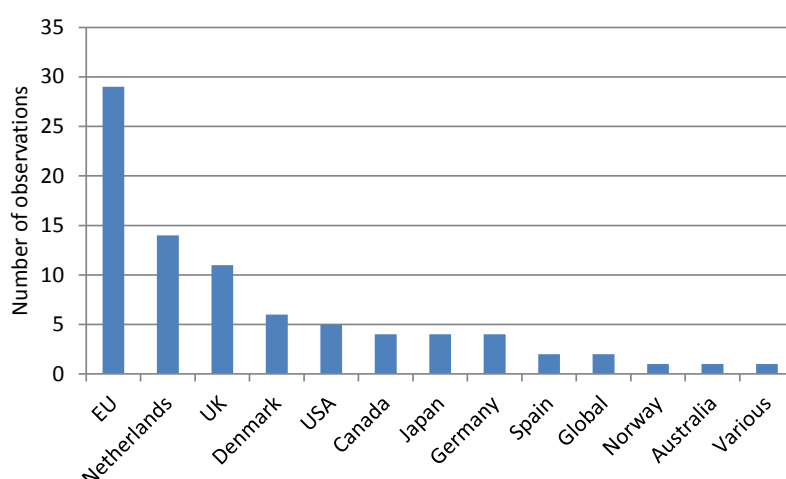


Figure 1 Country or continent of origin of the cost estimates

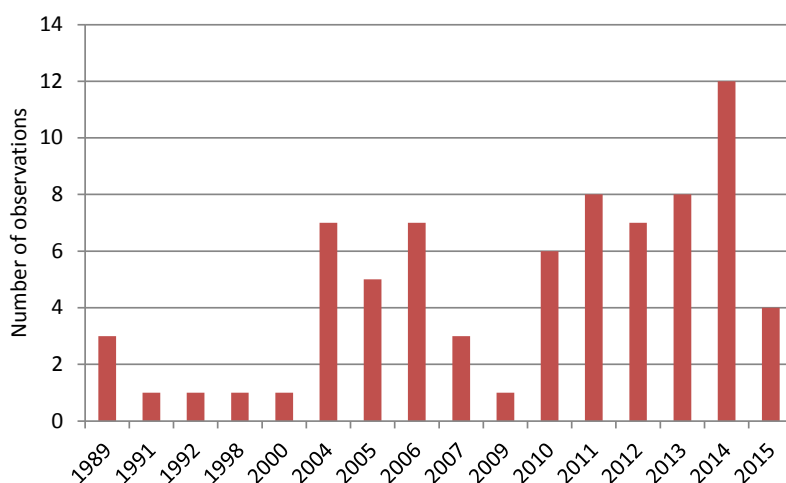


Figure 2 Distribution of the cost estimates over the study years

Table 2 presents the mean costs in increasing order per substance and shows that the costs related to D4/5 are on average lowest (EUR 75 per kg) and related to PFOS highest (EUR 1.2 million per kg). The differences in mean cost values are statistically significant (Kruskal-Wallis test statistic is 25.115,  $p < 0.01$ ). However, the spread in the values retrieved from the 36 studies included in the database is high as can be seen from the minimum and maximum cost values, the standard deviation and also when examining the column with the median values. Here we see that the ordering of the cost-effectiveness related to the first six substances changes if we use the median instead of the mean value. PFOS remains the most expensive to eliminate using both the mean and median value. Moreover, with the exception of the 4 observations for PFOA, the median value for the other substances is many orders of magnitude lower. Excluding PFOS, the median cost is maximum EUR 1507/kg, whereas, the associated mean value is close to EUR 46 thousand/kg. The number of observations is very limited for PFOA, where nevertheless mean and median costs per kg are almost the same. Most cost figures were obtained for PFOS, the most expensive substance to eliminate, i.e. 23 observations from 8 different studies. The average number of observations from the collected studies is 2 and varies between 1 and 6.

### 3.2 Substitution, emission reduction and clean-up

As indicated above, a distinction has to be made between different types of costs. We make a broad distinction between the costs of substitution, emission reduction and remediation/clean-up. The different cost types across the different substances are presented in Figure 3. The two highest values for PFOS are excluded in this figure in order to keep the Figure readable.<sup>4</sup>

Cost estimates for **substitution**, i.e. replacing a PBT by another substance (or introducing a PBT-free process) are often relatively low (and sometimes even negative<sup>5</sup>).

We did not find many examples of cases in which the risk reduction measures for PBT substances consist of **emission reduction**, i.e. continuing the use of the substance (in 'closed' applications) while ensuring a (close to) zero level of emissions and exposure.<sup>6</sup> **Clean-up** or **remediation** costs, like substitution costs, can differ widely, although they will obviously never become negative. Relatively low clean-up costs were found for HCB and HCH, as well as for some PCB cases (a few euros per kg). Other PCB cases show much higher costs (exceeding EUR 10,000 per kg in some USA 'Superfund' cases and possibly even above EUR 100,000 per kg for PCBs in 'open' applications). The highest remediation costs (up to more than EUR 1 million per kg) were found for PFOS. We should emphasize that clean-up or remediation can take on various forms: it does not only include the removal of a substance from the environment (e.g. soil or water), but also from products and man-made structures in which it is contained (e.g. PCBs from transformers; PFOS from firefighting equipment). In the latter forms, remediation comes close to substitution since the substance may be replaced by an alternative one. However, we still consider the costs of such replacement operations as 'remediation' costs, since they occur during the use stage of the product in which they are contained and not in the design or production stage.

<sup>4</sup> It should be emphasized, however, that the 'high end' values in our observations are still very relevant for the overall analysis since they are likely to reveal information on society's 'maximum willingness to pay'. See chapters 4 and 5.

<sup>5</sup> Negative and zero costs were only found for substitution, in 13% of the cases (n=6) related to substitution measures.

<sup>6</sup> In RPA (2004) emission reduction of PFOS as a temporary measure is discussed for one of the applications for which substitutes are not yet available, namely in the metal plating industry. The associated costs were estimated at GBP 3000 (EUR 4153) per kg emission reduction.



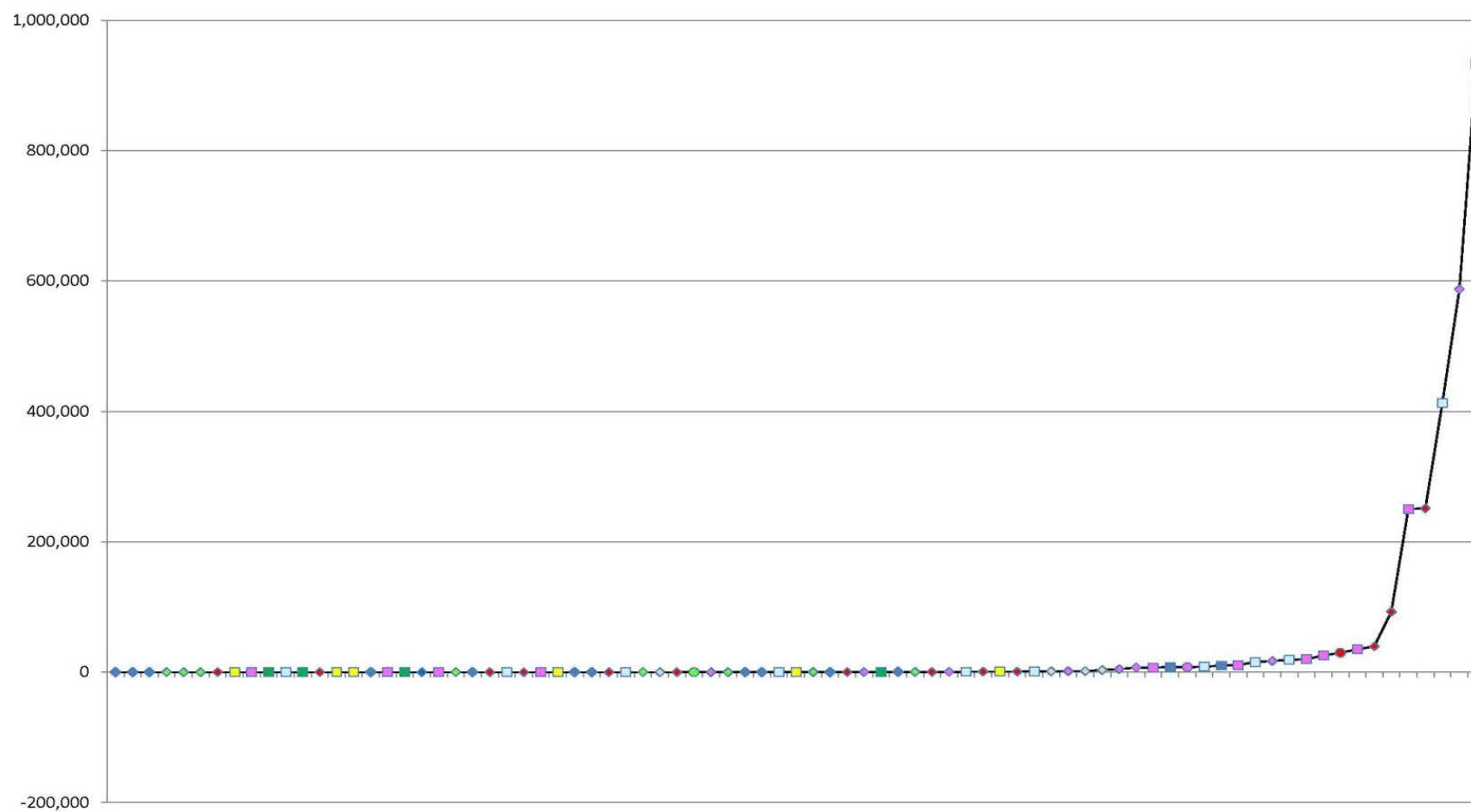


Figure 3 Unit costs (EUR/kg) below one million Euros<sup>1</sup> in increasing order for different substances and cost

**Explanatory notes:**

<sup>1</sup> Excluding two extremely high unit costs for PFOS (EUR 4.4 million/kg substitution costs and EUR 21.4 million/kg emission control costs)

Triangulars: substitution costs (n=45)

Squares: remediation costs (n=33)

Rounds: emission control costs (n=4)

Dark blue: HBCDD

Light blue: PCB

Dark green: HCB

Light green: D4/D5

Yellow: HCH

Red: d-BDE

Lila: PFOS

Grey: PFOA

In Figure 4 we present the mean costs for the different cost types. The number of observations is highest for substitution costs ( $n=46$ ), followed by remediation ( $n=33$ ). The number of observations for the unit costs of emission control is only 4. The total number of observations does not add up to 84 because one observation relates to both substitution and emission control and is therefore excluded from the analysis. The differences in mean unit costs are significant between substitution and emission control (Mann Whitney Z equals 1.967,  $p<0.049$ ), but not between substitution and remediation (Mann Whitney Z equals 0.885,  $p<0.376$ ) or remediation and emission control (Mann Whitney Z equals -1.565,  $p<0.127$ ). Median values are many orders of magnitude lower than the mean values due to the skewed distribution of cost figures with a long tail to the right and presented in Figure 5. The limited number of observations for emission control still generate the highest costs, but remediation costs are now as expected almost a factor 2 higher than the substitution costs (illustrating, despite the difference not being significant, the notion that it is generally cheaper to prevent than to cure). If we exclude the highest values for all three cost categories (EUR 21.4 million per kg for emission control, EUR 4.4 million per kg for substitution and EUR 933 thousand per kg for remediation), the mean unit costs for remediation are twice as high as for emission control while the substitution costs are slightly lower than the remediation costs (Figure 6). This clearly shows how sensitive the results are for outliers. Note, however that also the values excluding the outliers still have a wide spread, as can be seen from the error bars in Figure 6.

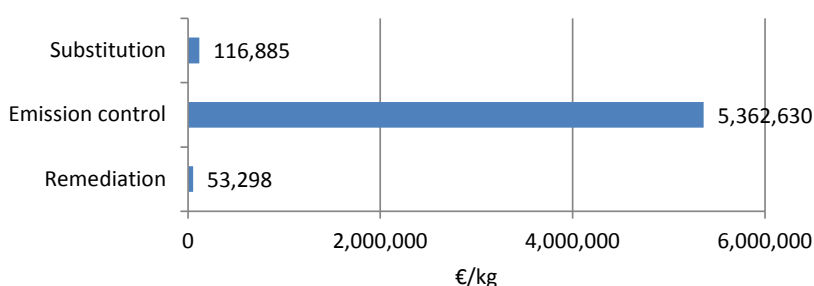


Figure 4 Mean unit costs across different cost types including outliers

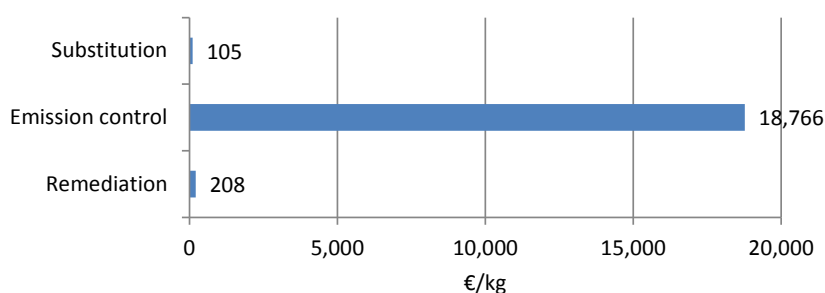
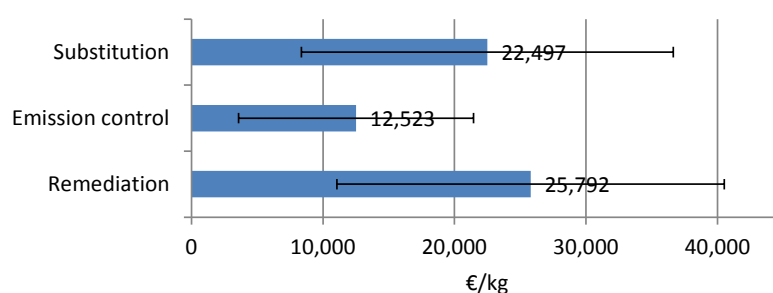


Figure 5 Median unit costs across different cost types



Note: error bars represent standard errors.

Figure 6 Mean unit costs across different cost types excluding outliers (highest unit costs per cost category)

### 3.3 Costs per kg avoided emission are higher than per kg avoided substance use

As noted above, if a PBT is currently already being used with reasonably adequate emission prevention, the cost of a future substitution will be much higher when expressed in euros per kg of emission avoided (or exposure reduction) than in euros per kg of substance replaced by an alternative. PFOS in photo imaging is a case in point: here the cost of per kg avoided emission is a factor 1000 higher than the cost per kg PFOS used, since only a small fraction of the PFOS used is eventually emitted.

The mean unit costs per kg avoided use and per kg avoided emission are presented in Figure 7. Over 40 percent of all observations relate to avoided use ( $n=36$ ). If we treat the remainder of the observations as avoided emissions (including remediation), these unit costs are almost a factor 15 higher than the unit costs of avoided use of substances.

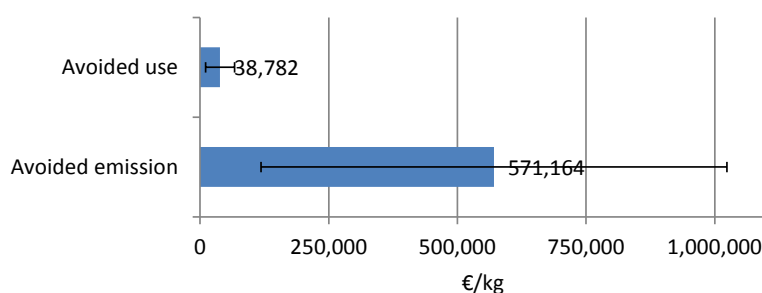


Figure 7 Mean unit costs per kg avoided use and per kg avoided emission

### 3.4 Inclusion and exclusion of cost components

The identified cases show a wide variety in terms of the cost types that are included in the estimate.

In the case of substitution, the simplest case is a ‘drop in substitute’ that can directly replace the PBT substance without the need for any process or equipment change and without any loss in quality or performance. In such cases, the only relevant cost is determined by the price difference between the two substances and the amounts needed. Our dataset contains some examples of such drop-in substitutes (e.g. EBP for deca-BDE or pFR for HBCDD), but such cases are relatively rare and usually only available for some specific applications of a specific substance. In many cases the substitution requires additional investments. An example is the substitution of EPS containing the flame retardant HBCDD by an alternative insulation material such as glass wool: this may require changes in the construction of the building. In other substitution cases, the result may not be equivalent to the original (e.g. D4/D5 substitutes in cosmetics, PFOA substitutes in certain textile applications and PFOS substitutes in metal plating). Moreover, the substitute may have its own environmentally harmful properties or create new health risks. The additional costs involved are sometimes hard to estimate and therefore often left out of scope. However, for instance in the case of D4/D5 there have been attempts to estimate the cost related to ‘performance reduction’. On the other hand, the substitute may also have a higher quality or certain advantages compared to the original (for instance in the case of certain insulation materials compared to EPS), and then the ‘real’ cost of substitution is lower than the price difference would suggest, since the substitute confers additional benefits.

Substitution also implies the need for product reformulation. Especially in the case of consumer products such as cosmetics, frequent product reformulation is common business (marketing) practice, and the relevant substitution cost is only the additional cost of ‘premature’ reformulation. Clearly, these costs may be reduced if industry is allowed more compliance time (as again the D4/D5 case shows).

In several cases, the replacement of a PBT substance implies the need to replace equipment as well, either specific capital goods for the production of the substance or equipment in which the substance was used or contained (e.g. transformers with PCBs). Whether or not such ‘sunk’ costs should be included in substitution cost estimates is a discussion that cannot be entered into here<sup>7</sup>, but it is obvious that their inclusion or exclusion affects the size of the total substitution costs.

<sup>7</sup> It is still a matter of discussion in ECHA (see ECHA 2013, section 3.2).



As far as clean-up costs are concerned, the main cost factor is usually related to the need to separate, isolate or remove the PBT substance from the product or the environmental compartment in which it is contained (or to separate the contaminated waste from other waste, as for instance in the case of flame retarded EPS in demolition waste). These costs can easily be orders of magnitude higher than the cost of collection, transport and final treatment or disposal of the substance (incineration, decomposition by other (bio)chemical processes, landfilling or controlled storage).

Restrictions on the use of a PBT substance may have implications for the opportunities to recycle the materials or products that contain these substances. Although this is usually acknowledged and in some cases (temporary) exemptions for recycling are made, the societal cost of foregone recycling opportunities is generally not included in cost estimates of restriction proposals. Norden (2015) indicate that half of the costs of recycling, which they estimate at EUR 800 – EUR 1000 per tonne for LCD TV's, can be recovered directly through enhanced material value and that avoided costs for incineration and/or landfill, if the need for these disposal routes are reduced, should be taken into account as well.

Sometimes indirect costs occur that can further add to the total cost of PBT substance risk reduction. An example is the Barendrecht railway tunnel PFOS case, where full PFOS removal might require the replacement of the entire firefighting system, implying the need to close the tunnel temporarily, with associated economic losses. Wider economic impacts were also included in the estimate made of the impact of imposing restrictions on the use of D4, D5 and D6 in Canada.

### 3.5 Amounts and concentrations; economies of scale; 'closed' and 'open' applications; 'point' and 'non-point' sources

The collected evidence confirms the intuitive expectation that it will become more costly to get rid of PBT substances (on a per kg basis) as the amounts become smaller and the concentrations lower. Economies of scale play an important role here. As long as a substantial stock of pollutants is located at a specific site under controlled conditions (as in the Australian HCB case) the remediation cost per kg pollutant is relatively low (in this case about EUR 5 per kg), even if the total amount of money can be substantial. In the New Bedford PCB remediation case the three technologies for remediation of the 'hot spots' (with PCB concentrations in sediment between 2000 and 5000 ppm) had a cost effectiveness between EUR 100 and 500 per kg, whereas the clean-up cost for the entire harbour area (concentrations up to 50 ppm) was estimated at about EUR 3300 per kg.

The pattern of increasing cost with decreasing concentrations is also clear in one of the PFOS cases (Barendrecht), where an initial clean-up appeared to be insufficient to remove the PFOS completely. The additional cost of a second round of clean-up, to achieve negligible PFOS residuals, would be between 10 and 400 times higher per kg of PFOS removed.

In order to test the impact of the amount of kg's on the unit costs and see if we can detect any economies of scale, we carried out a simple correlation analysis, relating the log transformed unit costs to the log transformed kg's released, controlled or remediated. Additionally, a regression analysis was performed to see if we can explain the variation in the observed unit costs across the different substances and cost types, including also control for other possible influencing factors, such as study year and study location. The correlation between the unit costs and kg's of the substances removed is highly significant and negative ( $r=-0.723$ ,  $p<0.001$ ), suggesting that the

unit costs decrease as the amount of the substance is replaced, its emission avoided, or removed. A similar significant negative effect is found in the regression analysis presented in Table 3 below. The double logarithmic form of the relationship between the dependent variable (natural log of the unit costs per kg) and the independent variable (natural log of the kg's removed) implies that the coefficient estimate can be interpreted as an elasticity: a 1% increase in the elimination of the substances (in kg's) results all else being constant in an almost similar decrease of the unit costs (EUR/kg) of 0.98%. Besides the constant term, none of the other explanatory factors are statistically significant except for the substitution costs, which are significantly lower than the remediation costs<sup>8</sup>, which are the baseline category in the regression model. Although the model fit is good and highly significant (the estimated model explains more than half (52%) of the observed variation in the unit costs per kg), the number of observations is very small and not equal to 84 due to the fact that in a lot of studies information about the amount of the substance removed is missing, and only unit costs are given. The results presented here therefore have to be interpreted with the necessary care.

*Table 3 Results of the linear regression analysis (natural log of the unit costs per kg is the dependent variable)*

Explanatory factor	Coefficient estimate	Standard error
Constant	12.950**	2.289
Amount of substance (ln(kg))	-0.976**	0.252
Substance (baseline is PFOS/PFOA)		
D4/5 (dummy)	-1.676	2.732
HCH/HCB (dummy)	-1.080	2.448
HBCDD (dummy)	1.797	2.211
d-BDE (dummy)	-0.410	1.813
PCB (dummy)	-0.794	2.325
Cost type (baseline is remediation)		
Substitution (dummy)	-3.262*	1.503
Emission control (dummy)	-1.851	2.859
European study (dummy)	-0.686	1.686
Study year (0-26)	-0.100	0.143
Model summary statistics		
F statistic	5.537**	
Adjusted R-square	0.519	
N	42	

\*  $p < 0.05$ ; \*\*  $p < 0.01$

A final comment is that once control is included for other influencing factors, in particular significant factors such as the size of substance removal and the type of costs (avoided use or avoided emissions), the average unit costs per kg seem transferable across substances, i.e. generally applicable to the various substances, in view of the fact that the type of substance does not have a significant effect on the

<sup>8</sup> This seems to contradict the finding in section 3.2 that the difference in mean unit costs between substitution and remediation was not statistically significant. This can be explained by the difference in type of analysis applied. In the regression analysis, the impact of various factors (including the type of measure) on the costs per unit is tested simultaneously, whereas in 3.2 the mean values of the costs per unit for the two types of measures were compared.

mean unit costs, irrespective of the baseline category, i.e. the same results are found when using other substances as the baseline category. Hence, differences do not occur anymore between substances once we control for (1) the amount of the substance and (2) whether the unit costs relate to substitution costs or remediation costs.

In line with the observation that unit costs increase with decreasing amounts and concentrations, clean-up of (ground)water pollution appears to be generally more expensive than soil pollution (illustrated by the Dutch HCB, HCH and PCB sanitation cases and the Arnsberg PFOS case in Germany, see the respective Annexes). This is probably due to the much wider dispersion and lower concentrations in (ground)water and the longer time it takes to extract the substance from this medium. Furthermore, costs tend to increase over time if the pollutant remains uncontrolled and gets diluted in the environment (which will happen more readily with relatively 'mobile' substances such as PFOS compared to less mobile substances that bind to the substrate).

Likewise, there is a clear difference in cost levels between 'closed' and 'open' applications, i.e. on the one hand situations in which the substance is contained in a relatively compact product or object (e.g. PCBs in transformers and capacitors) and on the other hand situations where the substance is spread over a relatively large surface or area (e.g. PCBs used in paint and other construction materials, HCH residues used for road paving, PFOS in sewage sludge used as a fertilizer). Related to this is the distinction between 'point' (concentrated) and 'non-point' (diffuse, scattered) sources. At 'point' sources the substance is located or used in relatively high amounts at a limited number of sites (e.g. industrial uses of PFOS), whereas with 'non-point' sources the substance is widely dispersed, in relatively low quantities, among numerous sites (for instance deca-BDE in TV sets). Normally, the substitution or remediation cost will be relatively low with 'point sources'. However, specific geographic circumstances can lead to high clean-up costs for 'point sources' as well, as illustrated by the Sparrevohn PCB case in the USA (remote location).

## 4 Decision making on PBTs

If we want information on the cost effectiveness of measures to control PBT substances to play a role in the development of a proportionality benchmark, we also need information on the decisions that have been taken concerning such measures. After all, the basic presumption of this study is that we can use information on the cost that society spends to avoid the environmental presence of and exposure to PBT substances can function as an (albeit imperfect) indicator/proxy for its willingness to pay to achieve this.

Clearly, in order to find indications for an upper limit we would want to have examples in which decision makers considered the cost of PBT substitution or removal to be too high. The cases reviewed in this study contain hardly any explicit statements concerning such 'disproportionate' costs. Exemptions from a ban on a specific substance, for example, are often motivated in general terms (pointing for instance at the non-availability of equivalent substitutes) without referring to any cost figures. But in some cases we can derive the implicit maximum willingness to pay by linking specific decisions to the estimated cost of that specific measure. Examples include the following (for details, see the case descriptions in the Annex):

- EU Directive 96/59/EC stipulates that Member States have to ensure that transformers containing more than 0.05 % by weight of PCBs are decontaminated. The cost of decontamination at this concentration level can be estimated between EUR 100 and 1000 per kg PCBs removed.
- In the USA, Superfund cases include a PCB remediation case where up to EUR 23,000 per kg of PCBs removed was spent, but also one case where an estimated EUR 16,000 per kg removed was considered too expensive.
- The cost of PFOS substitution in the application area photographic materials (which is exempted from the ban on PFOS under the EU POP regulation) was estimated between EUR 1500 and EUR 28,000 per kg emission reduction.
- In two Dutch distinct PFOS remediation cases (one concerning PFOS removal from a firefighting system and the other an accidental PFOS spill from such a system), costs of up to EUR 35,000 per kg removed have been made. In both cases, decisions have not yet been made on further remediation (at higher per kg cost).

From the limited available evidence we might conclude that there seems to be a rather broad 'grey zone' in which the cost of PBT substitution or removal can apparently be either proportionate or disproportionate, depending on the circumstances. In Figure 8 this is illustrated graphically. In the 'green' cost zone, measures to reduce the use, presence or emissions of PBTs will generally not be rejected for reasons of cost effectiveness. In the 'red' zone the cost tends to become prohibitive, or at least decisions tend to be postponed or (temporary) exemptions are granted. In the grey (or rather 'mixed red-and-green') zone the proportionality will be (co-)determined by situation-specific conditions and considerations. For illustrative purposes, the lower and upper margins of this 'grey zone' are put in the order of magnitude of EUR 1000 and EUR 50,000 per kg respectively. We should emphasize that this is just a general impression based upon a limited number of observations. Further expanding the evidence base could be helpful in confirming or adjusting the margins of the 'grey zone' and to refine it, for instance by distinguishing between cost types (substitution,

emission reduction or remediation costs)<sup>9</sup>, and between costs expressed per kg avoided use/presence (stock) or per kg avoided emission (flow).



Note: the red and green areas do not relate to actual cases and are for illustrative purposes only.

**Figure 8** Graphical representation of the principle of establishing an 'orders-of-magnitude' zone where the costs of PBT measures (in EUR/kg) may (green) or may not (red) be acceptable for cost-effectiveness reasons

<sup>9</sup> This does not imply that we recommend different benchmarks for different types of measures.

## 5 Towards a benchmarking approach in proportionality assessment

Now that we have collected a number of cost estimates for the various types of costs related to PBT control, and got an impression of the variation in their level and the factors that may explain this variation, we can go back to the question how useful this information is for the purpose of our study and what its relevance is for a possible benchmarking approach in the proportionality assessment for PBTs under REACH.

First of all, it seems clear that the broad range of cost effectiveness estimates, covering a wide range of rather specific situations concerning PBT substitution and remediation, does not provide much direct clues for a single benchmark that could be used to assess cost effectiveness of measures addressing PBTs. Even though the upper margin of the 'grey zone' mentioned above gives some idea of the order of magnitude where the likelihood of disproportional cost tends to become substantial, it can certainly not be used as a 'pass-fail' criterion in decision making.

Perhaps the main lesson from this study concerns the importance of distinguishing between different types of costs. In particular, a distinction should be made between cost effectiveness estimates relating to the amount of substance used or present in products/installations and estimates relating to the amount emitted to (or removed from) the environment. For example, if the cost of a substance ban is estimated at X euros per kg of avoided emission, it should not be compared with the cost of another substance ban if that cost was expressed per kg of avoided substance use.

In addition, it will only make sense to use existing cost effectiveness estimates as a reference if these estimates are 'complete', i.e. if they include all relevant cost components. For example, substitution costs that only include the material cost of the new substance are unlikely to be complete unless it is a 100% equivalent 'drop-in' substitute. Likewise, the result of the measure considered or proposed should be comparable with the result in the reference cases. However, the measures to achieve this result can be different. For example, cost effectiveness data on PBT clean-up from the environment can be used as a reference for the proportionality assessment of substitution or emission reduction measures, provided that the cost of the latter is expressed in euros per kg emission reduced/prevented.

Furthermore, it is obvious that in the actual assessment of proportionality a number of other considerations will be relevant, besides the cost-effectiveness in comparison with a possible benchmark. These may include, among others, the specific properties of the substance as well as socio-economic considerations (such as affordability and competitiveness) that are not accounted for in the cost effectiveness estimates.



## 6 Conclusions and areas for further work

This study aimed at setting some initial steps towards benchmarking for the proportionality assessment of PBT restriction proposals and authorization applications by gathering information on the cost effectiveness of (policy) measures that have been proposed and accepted or rejected to restrict the use and emissions of and exposure to these substances. As a general conclusion, one might say that the information gathering has led to a substantial evidence base in terms of cost effectiveness data. However, the role that these data play or have played in decision making is much less clear, as explicit statements on 'disproportional costs' are scarce. The steps taken should therefore indeed be considered as 'initial' in the sense that there is probably still quite a way to go until the envisaged benchmarking.

The cases analyzed in this study show a wide range of cost effectiveness estimates for PBT substitution, emission reduction and remediation. The evidence also suggests that substitution is usually (but not always) less expensive than remediation. Furthermore, substitution costs are usually much higher when expressed in EUR per kg emission avoided than in EUR per kg of substance replaced. The additional cost of the alternative substance (in case of substitution) and the cost of final disposal (in the case of remediation) tend to be relatively small components of the overall cost. The main (differences in) costs are related to other (case/situation specific) factors. In clean-up and remediation cases, these factors often include the efforts that are needed to separate the PBT from the product, installation or environmental media in which it is present. Especially when the concentrations of the PBT become very small, the costs per kg removed tend to increase significantly.

The comparability of cost effectiveness estimates is hampered by the differences in the cost types that are included or excluded, as well as in the assumptions that are made (which is often inevitable due to lacking data, e.g. on amounts used and on emission factors). Furthermore, a distinction has to be made between the cost per kg of 'stocks' (amounts used or present in products, installations or the environment) and per kg of 'flows' (emissions). Obviously, treating PBTs as a single category also disregards the differences in properties of the various PBTs.

As indicated above, decisions on PBT substitution or remediation are often not explicitly referring to the associated cost. This further complicates the challenge of estimating society's 'maximum willingness to pay' to reduce the use of PBTs or their presence in the environment.

Despite these limitations and qualifications, the (very limited) evidence from our cases suggests an (as yet very wide) 'grey zone' (with margins the order of magnitude somewhere between EUR 1000 and EUR 50,000 per kg PBT substituted, remediated or reduced emission). Within this grey zone, measures may apparently be either proportionate or disproportionate from a cost effectiveness perspective. Clearly, the 'grey zone' should be further substantiated (and possibly narrowed down) by means of additional case studies and analyses. If consensus could be achieved on the existence of such a 'grey zone', including its margins, then it could fulfil a role in a benchmarking process, for instance in the sense that:

- proposals for substitution or clean-up with cost estimates below the lower margin would normally not be rejected for reasons of cost-effectiveness;
- proposals for substitution or clean-up with cost estimates above the upper margin would normally be rejected for reasons of cost-effectiveness;



- proposals for substitution or clean-up with cost estimates within the ‘grey zone’ would require a detailed assessment on a case-by-case basis, making sure that all considerations that play a role (besides cost-effectiveness) receive due attention .

With a view to further work in this area, the following recommendations can be made:

- In order to ensure that benchmarking is based on comparable grounds, one should ensure that cost estimates are complete (all relevant cost types included), specify the assumptions made and whether the estimates are expressed in EUR per kg of use/presence reduction ( in products, installations or the environment) or in EUR per kg of emission reduction.
- Since the assessment of proportionality is not the same as a dichotomous ‘pass/fail’ test on a single cost-effectiveness criterion, the additional considerations that may play a role could be specified explicitly.
- Within the framework of the present study an Excel database was prepared containing the main information for the cases presented in the Annex. For its future work SEAC (and others) could benefit from further expanding and developing this database, and keeping it up-to-date.
- It could be investigated whether the benchmarking approach, for which this study aimed to provide some building blocks, is possibly also applicable to other (non-PBT) substance categories.
- Clearly, the cost effectiveness estimates on which this study reports do not represent cost effectiveness curves showing the amount of each PBT that can be substituted/removed at a certain cost level. In many cases, the information needed to construct such curves was partly lacking. Attempts to construct such curves have been done in previous work (Environment Agency, 2011), and new efforts in this direction (building, among others, on information gathered in the present study) could be useful for SEAC’s work.

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## ANNEX: Substance specific information

### D4/D5

#### Introduction

D4 (octamethylcyclotetrasiloxane) and D5 (decamethylcyclopentasiloxane) are widely used in cosmetics and maintenance products, among others under the name cyclomethicone. The ECHA PBT Expert Group concluded in November 2012 that both D4 and D5 meet the Annex XIII criteria for a vPvB substance in the environment, and that D4 also meets the criteria for a PBT substance.

#### Substitution costs

Publicly available data on substitution costs for D4 and D5 is scarce. Lassen *et al.* (2005) presented some estimates of prices of alternatives for siloxanes in cosmetics. Prices range from half the price of D4/D5 (which was around DKK 45 or EUR 6 per kg) to double that price, which would imply substitution costs **between EUR -3 (negative costs) and EUR 6 per kg** of D4/D5.

The high estimate from Lassen *et al.* (2005), i.e. an increase in the price of the raw material of 100%, is also used in the draft restriction dossier (ECHA, 2015a), but with an estimated unit price for D4/D5 of EUR 4 per kg. On top of this, the draft restriction dossier estimates the net cost of reformulation, i.e. the additional cost incurred by industry due to the need to replace existing products earlier than planned. For the target product group (wash-off personal care products) the annualized replacement costs are estimated between EUR 4 and 38 million under a 5 year compliance period, and between EUR 20 and 58 million under a 2 year compliance period. Given an annual amount of 855 tonnes (823 tonnes D5 and 32 tonnes D4) to be replaced, this means reformulation costs ranging from almost EUR 5 to EUR 68 per kg D4/D5. Together with the additional cost of raw materials, this implies a substitution cost of **EUR 9 to EUR 72 per kg** of D4/D5 *substituted*. In the draft restriction dossier (Table F.7) the per kg estimate refers to the cost per kg *emission reduction* and is therefore higher: **EUR 38 to EUR 308 per kg** D4/D5. The dossier also attempts to estimate the welfare loss due to 'product performance reduction' as a result of the substitution. Including these estimates would further increase the substitution costs to a level between **EUR 264 and EUR 533 per kg** of D4/D5 emission prevented.

The report by Oxford Economics (2008) on the economic importance of siloxanes in Canada does not estimate substitution costs, but the expected loss in Canadian GDP if D4, D5 and D6 would be designated as 'CEPA-toxic' (CAD 312 mln). Given an estimated 4,085 tonnes of D4, D5 and D6 sold in Canada in 2007/8, this would mean a cost (in terms of GDP loss) of CAD 76 or **EUR 56 per kg** substituted.

Environment Canada (2010a,b) released two cost-benefit studies for possible regulatory options to control D4 and D5 in Canada. Option A would prescribe concentration limits for industrial effluents, whereas option B would impose limits on the D4 and D5 content in consumer products. The results as far as costs and emissions are concerned are presented in the table below. The releases mentioned are the sum of D4 and D5 releases from industrial plants and from consumer products and are before treatment in public wastewater treatment plants. The information on quantities of D4 and D5 used were removed from the reports, since they were considered to be confidential business data.

Scenario	Reduction in releases (tonnes) compared to base case		Net present value of costs (CAD mln)
	year 2012	year 2037	
A: regulating industrial releases	75	131	130
B: regulating consumer products	108	205	151

Assuming, for sake of simplicity, a linear increase in the development of emission reduction between 2012 and 2037, the total reduction over this period would be 2678 tonnes under scenario A, and 4069 tonnes under scenario B. The cost per kg D4/D5 emission reduction (expressed in net present value) would then be CAD 49 (EUR 36) in scenario A and CAD 37 (EUR 27) in scenario B.

#### *Inclusion and exclusion of cost categories*

The estimates by Lassen *et al.* (2005) relate to the price of the substituting substance. The draft restriction dossier also includes estimates for the reformulation costs (see above). The estimates in Environment Canada (2010a,b) include capital and operating costs (scenario A) and substitution and reformulation costs (scenario B). The estimate for Canada by Oxford Economics (2008) has an even wider scope, including indirect impacts on the economy as well.

#### *Functionality and performance issues*

Lassen *et al.* (2005) note that siloxanes have a number of properties which are not easily matched by alternatives. This 'performance reduction' is included in the high-end cost estimates presented in the draft restriction dossier (see above).

#### *Factors affecting estimated cost levels*

The substitution costs estimated by Lassen *et al.* (2005) are determined by the market prices of the substitutes, and therefore presumably largely by the production costs of these alternatives. These costs are only estimated for the use of siloxanes in cosmetics. The authors note that the use of alternatives will in general not require changes in production equipment. As noted, the substitution costs as estimated in the draft restriction dossier (ECHA, 2015a) also include reformulation costs. Since frequent reformulation is standard practice in the personal care products industry, only the additional reformulation costs are relevant that are due to the need to change the product ahead of the normal schedule. An important factor is therefore the compliance period: the time given to industry to phase out the use of D4/D5. As indicated above, the reformulation costs are substantially lower under a 5 year compliance period than under a 2 year compliance period.

#### *Evidence on cost considerations in decision making*

For the use of D4/D5, no legal restrictions or bans are in place as yet. The UK has recently submitted a draft restriction proposal under REACH (ECHA, 2015b).

The Risk Management Approach as proposed by Environment Canada and Health Canada (2009) refers in its socio-economic considerations (section 7.3) to Lassen *et al.* (2005). In order to prevent or minimize releases of D4 and D5 to the aquatic

environment, the Government of Canada identified two potential options to reduce the release of D4 and D5 to water:

- limit the quantity or concentration of D4 and D5 that may be contained in certain personal care products and, where appropriate, in other consumer products that are manufactured in and imported into Canada; and
- prevent or minimize releases to the environment from industrial users of these substances.

Neither of these regulatory options was eventually put in place.<sup>10</sup> The proposed regulation for D4 was replaced by a pollution prevention plan. D5 is no longer considered as a toxic substance by Environment Canada (Fishlock, 2011). Clearly, a reassessment of risks rather than cost effectiveness considerations has led to these decisions.

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<sup>10</sup> Source: Personal communication from Rosemary Leslie (Environment Canada) to Martien Janssen (RIVM), 1 June 2015.



## DecaBDE

### Introduction

DecaBDE (decabromodiphenylether; also known as bis(pentabromophenyl)ether) is used as an additive flame retardant, mainly in plastics and textiles. Potential breakdown products of decaBDE were identified as a PBT and vPvB according to the REACH Regulation and decaBDE was included in the Candidate List on 19 December 2012. An Annex XV restriction report on decaBDE was published by ECHA on 1 August 2014 (ECHA, 2014).

### Substitution costs

In the Annex XV Restriction Report (ECHA, 2014, building on RPA, 2014), the cost effectiveness (cost of reducing 1 kg of decaBDE emission) is estimated to be 464 EUR/kg. The cost-effectiveness of the proposed restriction for decaBDE is in the same order of magnitude (or lower) as previous restrictions under REACH on mercury, which, according to the Restriction Report, “has some similar environmental properties”.<sup>11</sup> Therefore, it is considered that the additional costs, due to the proposed restriction, are proportionate to the risk reduction. (p. 12-13). Cost estimates were based on the price difference between decaBDE and its alternatives, the ease in substituting decaBDE (drop-in alternatives will result in fewer costs than other alternatives) and the predicted future amounts of decaBDE used. A sensitivity analysis on the main parameters influencing costs was carried out (p. 13). To calculate the substitution costs and cost-effectiveness, it is assumed that the entire [EU] consumption of decaBDE would be replaced by ethane,1,2-bis-pentabromophenyl (EBP) both in textiles and plastics. This appears to be the most ‘convenient’ alternative as it does not require alteration of formulations and it is not much more expensive than decaBDE. For the purposes of the calculation of substitution costs, the price of decaBDE is estimated at EUR4/kg and the price of EBP at EUR4.5/kg. In other words, EBP is estimated to be **EUR 0.5/kg** (12%) more expensive than decaBDE. [...] <sup>12</sup>The cost-effectiveness estimates (cost per kg of avoided emission) vary between different applications from EUR30 per kg (outdoor textiles) to EUR770 per kg (indoor plastics). The reasons for this are the significantly higher emission factors for outdoor applications compared to indoors. On average, the cost of reducing one kg of decaBDE is estimated to be **EUR 464 per kg**. (p. 54-55). The sensitivity analysis shows that the cost-effectiveness may range between **EUR125 – EUR 4000/kg** (p. 58). It should be noted that these figures are per kg of decaBDE emitted.

In the report on the development of a method to construct abatement cost curves by the Environment Agency (2011), decaBDE was used as a case study. The most cost-effective option identified for deca-BDE is replacement with phosphorus flame

<sup>11</sup> The comparability of decaBDE and mercury has been questioned by stakeholders. The Background Document to the RAC and SEAC opinions on the restriction proposal (ECHA, 2015), comments to this statement: “However, the comparison of cost-effectiveness between decaBDE and mercury compounds is not straightforward as their individual circumstances (i.e. hazard potential / exposure in the environment), are not directly comparable. This precludes the use of the cost-effectiveness of previous restrictions as a benchmark of acceptable cost-effectiveness. However, this information remains relevant to a discussion on proportionality and is included as supporting information.”

<sup>12</sup> The concentration of EBP is assumed to be the same as the concentration of decaBDE (see Table 10 in ECHA, 2014).

retardant<sup>13</sup> in textiles. This option has a cost effectiveness of GBP 144 (**EUR 184 per kg**). With this option most of the deca-BDE emissions in the EU can be avoided (about 50 tonnes, mainly from textiles waste disposal).<sup>14</sup> The remaining options are much more expensive (ranging from GBP 22,659 to 195,620 or **EUR 30,500 to 263,000 per kg**) and each of them reduces EU wide emissions by just 80 kg per year. (p. 37-38). These expensive options include: replacement of deca-BDE with alternatives in polymers; replacement with other brominated flame retardants in HIPS to fire safety standard UK 94 V-1; replacement with halogen-free flame retardant; additional waste water treatment for polymer compounding and conversion; thermal oxidation of emissions to air for polymer compounding and conversion; and additional waste water treatment for textile compounding and application.

The cost estimates of decaBDE substitution by Lassen *et al.* (2006, Table 3.9) are expressed in terms of cost per kg of HIPS compound. HIPS with a different brominated flame retardant (BFR) was EUR 0.10 to EUR 0.40 more expensive than HIPS with decaBDE. For halogen free flame retardants the price difference was EUR 0.80 to EUR 1.10 per kg of compound. Elsewhere in Lassen *et al.* (2006, section 6.3) an average decaBDE content of 12% is mentioned for plastics in electric and electronic equipment in which this flame retardant is used. This points to substitution costs of **EUR 0.80 to EUR 3 per kg** of decaBDE for BFRs and **EUR 7 to EUR 9 per kg** of decaBDE for halogen free flame retardants.

LCSP (2005) presented cost estimates for decaBDE substitution in TV enclosures. Compared to HIPS with decaBDE the substitutes were estimated to be USD 0.18 to 0.99 per pound more expensive, which equals EUR 0.36 to 1.96 per kg. As in Lassen *et al.* (2006), the estimated decaBDE content is 12% by weight (LCSP, 2005, p. 20), which means substitution costs of **EUR 3 to EUR 16 per kg** of decaBDE.

### *Inclusion and exclusion of cost categories*

According to the Annex XV restriction report (ECHA, 2014), the available information supports the assumption that EBP is a drop-in substitute for decaBDE allowing cost calculation simply based on prices and concentration, excluding R&D and potential changes in the process.

### *Functionality and performance issues*

A potentially important issue in the assessment of equivalence of alternatives in the case of fire retardants is the extent to which the alternative's performance in terms of fire prevention is as good as that of the original substance. In the case of decaBDE, this seems to be less of an issue, as EBP is seen as a substance that can replace decaBDE from both a technical and an economic perspective (ECHA, 2015). However, there are concerns related to potential PBT/vPvB properties of EBP, which are currently being addressed by the ongoing Substance Evaluation under the REACH Regulation .

### *Factors affecting estimated cost levels*

In the Annex XV restriction report (ECHA, 2014) a single value is used for the substitution cost in all applications of decaBDE. The variety in the cost-effectiveness (in EUR per kg emitted) is explained by the differences in emission factors for different

<sup>13</sup> The exact name of the substitute is not specified.

<sup>14</sup> It needs to be highlighted that the cost-effectiveness estimates of this study are not comparable with the values in the restriction report by ECHA, e.g. due to much higher assumption on emissions factor from disposal on textiles (source: personal communication, Kalle Kivelä. ECHA, 1 June 2015).

applications. Likewise, in the abatement cost curve case study on decaBDE (Environment Agency, 2011) the cost estimates are strongly influenced by the assumptions regarding the emissions that can be prevented. High per kg costs are found for options that reduce EU wide emissions by just 80 kg per year each.

### Evidence on cost considerations in decision making

RPA (2014) lists the following existing and proposed legislation with restrictions on the use of decaBDE:

- Directive 2002/95/EC (RoHS), requiring that new electrical and electronic equipment (EEE) placed on the market shall not contain polybromodiphenylethers (PBDEs), including decaBDE, in concentrations higher than 0.1% w/w in homogeneous materials<sup>15</sup>;
- the Water Framework Directive, in which PBDEs<sup>16</sup> are considered as priority substances;
- REACH, in the context of which decaBDE is considered a “substance of very high concern” (SVHC);
- the Stockholm Convention: Norway submitted a proposal to include decaBDE in Annexes A, B and/or C, requiring the consideration of its global consumption, persistence, potential for long-range transport and adverse effects.

Cost considerations apparently have not played any role in the ban on PBDEs in EEE under Directive 2002/95/EC. In its proposal for this Directive (COM(2000) 347 final) the Commission argued that the presence of PBDEs in EEE was an impediment to recycling, since there is a risk of dioxins and furans being generated during extrusion<sup>17</sup>, which is part of the plastic recycling process. In addition, high concentrations of PBDEs had been found in the blood of workers in recycling plants<sup>18</sup>, whereas scientific observations indicated that PBDEs might act as endocrine disrupters. However, a risk assessment in 2004 suggested there was no need to restrict the use of deca-BDE and in October 2005 the Commission decided to exempt decaBDE from the ban. In 2008, however, this exemption was annulled by the European Court of Justice.

In the United States and Canada the use of decaBDE is being phased out under voluntary agreements between industry and authorities. Four states in the US had already banned the use of decaBDE some years ago (BSEF, 2010). The cost considerations underlying these bans seem to have a rather general character, pointing to the availability, feasibility and (sometimes) affordability of alternatives rather than to detailed quantitative cost analysis. For example, the state of Washington presented a PBDE Chemical Action Plan in 2006 (Washington State, 2006), stating:

*“[The Washington State Department of] Ecology conducted a Cost Benefit Analysis (CBA) of a statewide ban on Deca-BDE in electronic enclosures in order to weigh the benefits to human health and the environment against the costs to business. Information on*

<sup>15</sup> Directive 2002/95/EC has meanwhile been replaced by Directive 2011/65/EC, which contains the same ban on PBDEs in EEE (without exemptions).

<sup>16</sup> This relates only to tetra, penta, hexa and heptaBDE; not to decaBDE.

<sup>17</sup> It has been questioned whether this risk actually exists. We will not discuss the validity of this argument here, but simply note that it was used by the Commission.

<sup>18</sup> More recent publications on occupational exposure to DecaBDE give a mixed picture, suggesting that it may be mainly a problem in developing countries and countries in transition with inadequate (enforcement of) risk reduction measures. See e.g. Thuresson *et al.* (2006), Rosenberg *et al.* (2011), UNEP (2014).

*costs was hindered by difficulties getting information from businesses about their Deca-BDE use. Many businesses were reluctant to share cost data with us, possibly because the state could not provide confidentiality for this information. When it became apparent that critical data would not be available, Ecology developed an alternative model which we believe might be successfully used to compare costs to benefits. However, this model hinges on the identification of at least one safer, effective alternative to Deca-BDE, which has not yet been identified. In addition, there is considerable uncertainty in the data needed to quantify health benefits. Ecology is therefore unable to determine whether benefits exceed costs (or vice versa). Consequently, Ecology has concluded that the cost benefit analysis has limited utility at this time to inform decisions on phasing-out uses of deca-BDE."*

By the end of 2008, however, the Department of Ecology published a new report (Washington State, 2008) which concluded that for televisions and computers a safer and technically feasible alternative to decaBDE was available (namely resorcinol bis diphenyl phosphate, RDP). Pointing to the ban on decaBDE in EEE in the EU it assumed that "if these products can be made cost effectively and sold in Europe they can be made cost effectively and sold in the U.S. as well". For residential upholstered furniture, it was concluded that non-chemical design options were the safer, technically feasible alternative to Deca-BDE. These conclusions led to the ban on Deca-BDE in the mentioned applications as of 1 January 2011.

Norway has introduced a general ban on DecaBDE as of 1 April 2008, with an exemption for the transport sector (BSEF, 2010). Cost (effectiveness/benefit) considerations underlying this decision could not be found. The exemption was motivated by the 'big consequences' which a ban on decaBDE would have for the transport sector.<sup>19</sup> In 2013, Norway submitted a proposal to add decaBDE to the Stockholm Convention.

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## HBCDD

### Introduction

HBCDD (hexabromocyclododecane; also known as HBCD) is a flame retardant that has its main application in expanded polystyrene (EPS) and extruded polystyrene (XPS) used for insulation purposes in construction. In 2008 HBCDD was added to Annex XIV of the REACH Regulation due to its PBT properties. In 2013 HBCDD was added to Annex A of the Stockholm Convention.

### Substitution costs

ECHA (2013, Annex 5) presents the results of a case study on HBCDD. The use of alternative insulation materials to replace the use of HBCDD in EPS/XPS foams is the only measure considered. In the case of EPS, the alternatives generally have higher cost to achieve the same level of thermal insulation (ranges from **EUR 25 to 175 per kg** of HBCDD removed), whereas for XPS, the alternatives appear to have lower cost, giving a negative value for the cost per tonne of HBCDD use avoided (ranges from **EUR -67 to -187 per kg** of HBCDD removed). These costs only include the difference in prices between HBCDD-based products and the alternatives. They do not include the (potentially significant) cost implications that could occur as a result of the lost market, and lost residual value of capital equipment, for the current suppliers of HBCDD-based EPS and XPS to the insulation market.

In its two opinions on HBCDD authorization applications, ECHA (2015a and 2015b) assumed a 1 to 5% price increase of EPS with an alternative flame retardant (pFR) compared to EPS with the flame retardant HBCDD. On this basis SEAC estimated an additional price of **EUR 1.92 to 9.59 per kg** HBCDD substituted (Annex 1, Table B in both opinions).

The paper by Inoue *et al.* (undated) contains some cost estimates for an alternative flame retardant substance (tetrabromocyclooctane: JPY 650 or **EUR 3.40 per kg** more expensive than HBCDD) and for alternative insulation techniques (glass wool or polyurethane instead of polystyrene). Glass wool would increase the annual insulation costs in Japan by around JPY 2.5 billion, and polyurethane by around JPY 30 billion. Given an estimated annual use of HBCDD for insulation of 2000 tonnes, this means a substitution cost of **EUR 9 to 112 per kg** HBCDD.

Lassen *et al.* (2011) evaluated selected alternatives to flame retarded EPS used in the building sector: stone wool, polyurethane foams (PUR/PIR), wood fibre boards and cellular glass. The price of the cheapest alternatives ranges from more or less the same price as for flame retarded EPS to approximately 30% more. Alternatives of significantly higher price exist, but these are typically used because they have some desired technical advantages and would probably not be the first choice substitutes for general application. Detailed price comparisons are presented in Tables 5.7 – 5.9 of the report and summarized in the table below.

*Prices of alternative insulation materials (EUR per m<sup>2</sup> functional unit)*

Application	EPS/XPS	Glass/stone wool	PUR/PIR	Other*
External façade insulation	15	16 - 21		43 - 52
Flat roof insulation	13 - 24	22 - 48	16	59 - 87
Floor insulation	13 - 20		17 - 18	31 - 87

\* A.o. wood fibre, cellular glass.



These comparisons are per m<sup>2</sup> of functional unit (thickness of the material equivalent to EPS of 10 cm thickness). A cost estimate per kg of HBCDD can be derived as follows. The HBCDD content of flame retarded EPS and XPS ranges between 0.5 and 2.5% w/w (UNEP, 2011). EPS, which is less expensive and has the highest market share of the two, is at the lower side of this range (0.5 to 0.7% according to Lassen *et al.*, 2011). The density of EPS is between 15 and 35 kg/m<sup>3</sup> (Lassen *et al.*, 2011). One m<sup>2</sup> of flame retarded EPS with 10 cm thickness therefore has a weight of 1.5 to 3.5 kg and contains 7.5 to 24.5 grammes of HBCDD. Since the cheapest alternative in each application area is between 1 and 3 EUR per m<sup>2</sup> (functional unit) more expensive, the additional cost per kg of HBCDD substituted can be estimated at between **EUR 41 and EUR 400 per kg HBCDD**.

The study by UNEP (2012) contains information on a number of alternatives for HBCDD as a fire retardant (FR) in EPS and XPS, including some global cost estimates. It states: "Some Parties indicated in their responses higher costs of the Polymeric FR compared to HBCD. However, no financial values were included to support this. According to one producer of the Polymeric FR, manufacturing flame retarded products with the alternative to HBCD is not anticipated to have any significant impact on the cost competitiveness of EPS or XPS. It remains unclear whether the flame retardant represents a significant factor in the price of the final product (EPS/XPS insulation board). More precise cost estimates will not be available until the Polymeric FR is fully commercialized." (p. 14-15)

In a study concerning a possible ban on the use of HBCDD as a flame retardant in electrical and electronic equipment under the RoHS Directive (Umweltbundesamt, 2014, using data from Maag *et al.*, 2010) the additional material costs of the alternative (HIPS/PPE containing halogen-free flame retardants) were estimated at EUR 1.33 per kg of plastics. In addition, there would be investment costs of EUR 0.23 per kg of plastics. Given a 7% HBCDD content in the original HIPS this would imply substitution costs of **EUR 22 per kg of HBCDD**.

### *Inclusion and exclusion of cost categories*

The reported substitution cost estimates generally relate to the additional cost of using the alternative substance or material. In the case of drop-in substitutes, this is basically the price difference between the alternative flame retardant and HBCDD. In the case of alternative insulation materials the cost estimates usually relate to the same functional unit (amount of material needed to achieve the same level of thermal insulation as with EPS/XPS). As indicated above, they do not include the potential cost implications of market loss and capital destruction. The above mentioned cost estimates do also not include the cost of 'foregone recycling' in the case that there would be no exemption for HBCDD in recycled materials. Recent German reports (Mäurer and Schlummer, 2014; Fraunhofer IBP, 2014) have further investigated the issue of HBCDD in the existing stock of EPS insulation materials and its implications for recycling. However, they do not address the associated costs quantitatively.

### *Functionality and performance issues*

The drop-in substitutes identified in the various studies provide for an equivalent level of fire protection and do not affect the functional properties of the EPS/XPS. For the alternative insulation materials, cost comparisons relate to the amount of material needed to reach similar levels of thermal insulation and therefore also entail functional equivalence. However, they do not take into account the extent to which the alternative material is readily applicable (for example, the layer of mineral wool needed to reach the same level of insulation as with EPS/XPS has to be thicker, and the available space

for this may be lacking). Possible other disadvantages of the alternative (e.g. higher transport costs, life cycle energy requirements and environmental performance) are also not taken into account.

### *Factors affecting estimated cost levels*

Substitution costs for HBCDD in EPS and XPS used as insulation materials strongly depend on the type of substitution. If it is assumed that an equivalent drop-in flame retardant is available, the additional cost per kg of HBCDD replaced is relatively small. Cost estimates for substitution by alternative insulation materials vary widely, between negative amounts and hundreds of euros per kg HBCDD replaced. This variation is partly due to variability in the physical properties of the polystyrene (density, HBCDD content).

### *Costs of disposal, emission control and remediation*

Broomfield *et al.* (2010, chapter 5) performed a Cost Benefit Analysis for five scenarios for HBCDD control, focusing on the UK. Two types of cost are distinguished: product replacement costs and safe disposal costs. In all scenarios, the main cost is associated with separation of materials containing HBCDD. Scenario 1 (a complete ban on the manufacture or use of HBCD) would entail annualized costs of GBP 1.4 to 6.9 billion, and prevent the emission of 730 tonnes of HBCD per year by 2030, implying a cost of almost **EUR 3000 to almost EUR 13,000 per kg HBCDD**. Scenario 5 (no bans or use restrictions, but safe disposal (i.e. incineration) of materials containing HBCDD) would entail annualized costs of GBP 1.4 to 6.0 billion, and prevent the emission of 490 tonnes of HBCDD per year by 2030, implying a cost of about **EUR 4000 to EUR 16,000 per kg HBCDD**. Scenarios 2, 3 and 4 entail exemptions for certain applications and have comparable cost effectiveness estimates to Scenario 5.

### *Inclusion and exclusion of cost categories*

The main cost item in the estimates by Broomfield *et al.* (2010) is the cost of separating construction and demolition material as well as EEE waste. These costs account for more than 80% of total cost in all scenarios. Other costs under 'safe disposal' include the cost of incineration and regulatory costs. The 'product replacement costs' mainly include the losses (and gains) in turnover for the various products.

### *Control or destruction*

In all scenarios distinguished by Broomfield *et al.* (2010) the materials containing HBCDD are incinerated at the end of their lifetime. Broomfield does not assess the economic impact of incineration compared to recycling. Technical feasibility (effective screening techniques, availability of incineration capacity) is not taken into account in this study.

### *Factors affecting estimated cost levels*

The dominant cost factor in Broomfield *et al.* (2010) is the cost of separating materials containing HBCDD for safe disposal.

### *Evidence on cost considerations in decision making*

In 2013, HBCDD was added to Annex A of the Stockholm Convention. This means the substance must be phased out by countries that have ratified the treaty. A five year



exemption is allowed for its use in EPS/XPS building insulation materials. The risk management evaluation underlying this ban (UNEP, 2011) includes a section on economic aspects. Concerning substitution it contains some general considerations, without quantification. With respect to the need to destroy existing stocks of materials containing HBCDD, it notes that specialized waste management and disposal could be costly. It also gives some estimates of the volumes involved, but does not mention the monetary amounts presented in the Broomfield *et al.* (2010) scenarios. As far as the replacement of HBCD in HIPS in EEE is concerned, the report refers to Maag *et al.* (2010).

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## HCB

### Introduction

Hexachlorobenzene (HCB) was widely used as a seed dressing to prevent fungal growth on crops until 1965, and it was also a component of fireworks, ammunition and synthetic rubbers (Tong and Yuan, 2012). It is listed under Annex A and Annex C of the Stockholm Convention.

### Costs of disposal, emission control and remediation

HCB is often one among many pollutants at contaminated sites, which complicates the attribution of remediation costs to this specific substance. The HCB waste at Botany Park (Sydney, Australia) is an example of a case where HCB is the only pollutant of concern. Over the past 30-40 years several proposals have been put forward and rejected to deal with this legacy, either within Australia or by exporting the waste for incineration in Europe. By 2012, a decision on the destiny of the waste still had not been taken. The cost of remediation was estimated at AUD 95.2 million (EUR 66.8 mln at current exchange rate) in 2009, and expected to rise by AUD 18 million (EUR 12.6 mln) in 2010 (Trading Room, 2010). Approximately 15,000 tonnes of concentrated HCB (mixed and in various forms) and low level (packaging etc) waste is now stored in containers at Botany Industrial Park (Chappie, 2012), so given a total cost of almost EUR 80 mln this points at a cost of some **EUR 5 per kg** HCB.

Versluijs (2015) provided us with an estimate of the costs of about 10 locations in the Netherlands where soil and groundwater pollution by HCB has been remediated. Using a number of standardized assumptions concerning the size of the pollution he arrives at a range from **EUR 1.10 to 208 per kg** HCB (the lower end of the range relating to soil, and the higher end to groundwater).

EPA (2010) mentions Base Catalyzed Decomposition (BCD) as a non-combustion technology for POP remediation that is suitable for (among others) HCB. The technology has been used at full-scale in various countries around the world including Spain, Australia, Japan, Czech Republic and Mexico. The cost estimate of EUR 1,400 to 1,700 per ton (**EUR 1.40 to 1.70 per kg**) is for the year 2004 and is based on information from IHPA (2009).

### *Inclusion and exclusion of cost categories*

The Australian and Dutch cost estimates include the full cost of remediation in specific cases, whereas the estimate by EPA (2010) for BCD only includes the cost of decomposition of the substance by the specified technology.

### *Control or destruction*

In the Australian case, the remediation technology still remains undecided, although all remaining options in 2012 included some form of treatment (incineration or bioremediation). As for the Dutch cases reported on by Versluijs (2015), the soil pollution related ones involved destruction (thermal treatment), whereas for the groundwater pollution this remains unclear. The BCD technology (EPA, 2010) clearly involves destruction (decomposition) of the HCB.

### Factors affecting estimated cost levels

In the Australian case, the cost per kg is relatively low due to the highly localized and concentrated nature of the HCB waste. Nevertheless, the cost continues to increase as decisions on final treatment are postponed. In the Dutch cases, we can see a clear influence of the environmental compartment where the HCB is located, with soil giving significantly lower costs than groundwater (probably due to the much wider dispersion and lower concentrations in groundwater and the longer time it takes to extract the substance from groundwater).

### Evidence on cost considerations in decision making

No evidence has been found on the role of cost considerations in decisions on the clean-up of HCB contaminated sites. In the Australian case, the long lasting lack of action was mainly due to political resistance (e.g. public opposition to sending the HCB to Denmark or Germany for incineration), and not because the various clean-up options were considered 'too expensive'.

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## HCH / lindane

### Introduction

Hexachlorocyclohexane (HCH) was produced and widely used as an insecticide in the past. The three main isomers ( $\alpha$ -HCH,  $\beta$ -HCH and  $\gamma$ -HCH) are since 2009 listed in Annex A of the Stockholm Convention, which means that parties have to prohibit and/or eliminate the production and use, as well as the import and export of these POPs. For  $\gamma$ -HCH, also known as lindane, there is an exemption for its use as a human health pharmaceutical for control of head lice and scabies as second line treatment.

The main pollution legacies of HCH are due to the fact that since 1952 'pure' lindane ( $\gamma$ -HCH) was produced from mixtures of the isomers, leaving substantial amounts of  $\alpha$ -HCH and  $\beta$ -HCH as by-products without value. These stockpiles often remained at production locations and parts of them were dumped or used for road paving.

### Costs of disposal, emission control and remediation

Versluijs (2015) provided us with an estimate of the costs of about 40 locations in the Netherlands where soil and groundwater pollution by HCH/lindane has been remediated. Using a number of standardized assumptions concerning the size of the pollution he arrives at a range from **EUR 4.50 to 760 per kg** HCH/lindane (the lower end of the range relating to soil, and the higher end to groundwater).

Spuij and Urlings (1989) made a cost comparison between biological treatment and conventional methods for HCH, benzene and chlorobenzene from groundwater at the site of a former pesticides plant. The cost of biological treatment was estimated at NLG 250 or **EUR 113 per kg** pollutant. It should be noted, however, that the technology was not able to remove  $\beta$ -HCH, which is the most persistent isomer.

EPA (2010) mentions Base Catalyzed Decomposition (BCD) as a non-combustion technology for POP remediation that is suitable for (among others) HCH. The technology has been used at full-scale in various countries around the world including Spain, Australia, Japan, Czech Republic and Mexico. The cost estimate of EUR 1,400 to 1,700 per ton (**EUR 1.40 to 1.70 per kg**) is for the year 2004 and is based on information from IHPA (2009).

Vijgen (2006) presents quantitative cost data on actual cases of HCH remediation and also estimates of total global lindane production (and associated HCH waste that has been stocked or dumped). In the Netherlands, EUR 28 million has been spent on remediation sites where 1500 tonnes of HCH were dumped (part of this is still to be remediated), implying a cost of (at least) **EUR 19 per kg** HCH. In Spain (Basque country) EUR 50 million has been spent on sites where 82,000 tonnes were dumped. Of this amount, EUR 8.4 million was spent to treat 3,500 tonnes using the BCD process. The remaining was used for two safe hazardous waste landfills. This points at costs between (at least) **EUR 0.53** (landfilling: EUR 41.6 million for 78,500 tonnes) **and EUR 2.40** (BCD) **per kg** HCH.

### *Inclusion and exclusion of cost categories*

The cost estimates by Versluijs (2015), Spuij and Urlings (1989) and Vijgen (2006) include the full cost of remediation in specific cases, whereas the estimate by EPA (2010) for BCD only includes the cost of decomposition of the substance by the specified technology.

### Control or destruction

As for the Dutch cases reported on by Versluijs (2015), the soil pollution related ones involved destruction (thermal treatment), whereas for the groundwater pollution this remains unclear. The BCD technology (EPA, 2010, and part of the Basque case reported in Vijgen *et al.*, 2006) clearly involves destruction (decomposition) of the HCH. The same is true for the biological treatment as reported by Spuij and Urlings (1989). The main part of the Basque HCH legacy (with the lowest cost estimate) is landfilled and therefore not actively decomposed.

### Factors affecting estimated cost levels

The lowest costs are found in cases where the HCH is still stockpiled at the plant where it was produced. The costs become higher as the HCH is dispersed in the environment. In the Dutch cases (Versluijs, 2015), we can also see a clear influence of the environmental compartment where the HCH is located, with soil remediation giving significantly lower costs than groundwater treatment.

### Evidence on cost considerations in decision making

As the report by Vijgen (2006) shows, there is still a substantial amount of unremediated HCH stocks and contaminated sites. It seems quite probable that, at least within each country that has a HCH legacy, the lowest-cost options for sanitation have been carried out with priority (cherry picking). However, any explicit decisions to abandon remediation motivated by cost considerations could not be found.

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## PCBs

### Introduction

PCBs (polychlorinated biphenyls) were widely used until the mid-1970s, among others as a dielectric and cooling fluid in transformers and capacitors, as well as in a number of 'open' applications (such as paint, ceiling panels and sealants), in which PCBs were used to enhance physical and chemical resistance, to act as plasticisers, flame retardants, impregnating agents, coolants, and lubricants. In the EU the production, marketing and use of PCBs have been restricted since 1976 and prohibited since 1985. In 1996, Directive 96/59/EC introduced provisions for the environmentally safe disposal of existing PCB stocks. Member States were required, among others, to dispose of big PCB containing equipment (with a volume of more than 5 litres) by the end of 2010 at the latest. Other (equipment with) PCBs have to be disposed of 'as soon as possible', without a specified deadline.

### Costs of disposal, emission control and remediation

Most of the available sources on cost effectiveness data for PCBs relate to the removal of PCBs from electric and electronic equipment (mostly transformers and capacitors) and the subsequent destruction of the PCBs. Cost estimates are usually given in euros per kg of the product containing the PCBs. These estimates are generally in the order of magnitude of EUR 1 per kg of total weight.<sup>20</sup> For example, EC (2014) reports the following estimates:

- liquid PCB from a few ppm to pure PCB: 50-500 EUR/tonne;
- capacitors: depending on the size: 500-1 300 EUR/tonne;
- transformers: depending whether the core is in copper or aluminium and if the metallic parts can be recovered: 600-1 500 EUR/tonne.

The amounts mentioned under the first bullet imply a cost of **EUR 100 to 1,000 per kg** PCBs for mixtures with PCB concentrations of 0.05% (500 ppm, the lower limit for which disposal of PCBs is mandatory).

Vermeulen *et al.* (1989) give estimates of (subsidized) costs per kg of PCBs removed under an early subsidy scheme in the Netherlands: **EUR 6 per kg** PCBs for transformers and **EUR 101 per kg** PCBs for capacitors.

Versluijs (2015) provided us with an estimate of the costs of about 50 locations in the Netherlands where soil and groundwater pollution by PCBs has been remediated. Using a number of standardized assumptions concerning the size of the pollution he arrives at a range from **EUR 1.30 to 1,200 per kg** PCBs (the lower end of the range relating to soil, and the higher end to groundwater).

The main PCB related Superfund case in the USA is the New Bedford Harbor site (see Box 1). Another 'PCB only' Superfund case is the Sparrevohn Long Range Radar Station, Alaska. At this Air Force site, solvent extraction was used to remove PCB contamination in 1996. 33.8 kg of PCBs were removed at a total cost of USD 828,179, implying a cost of almost USD 24,500 or **EUR 23,000 per kg** of PCBs removed. The relatively high cost in this case can be partly explained by the remote location of the site, which can only

<sup>20</sup> For small capacitors this estimate of 1 EUR per kg was confirmed by the Dutch electronics recycling organisation Wecycle (e-mail from Mark Tilstra of Wecycle to Martien Janssen of RIVM, 18 February 2015).



be reached by air. The cost for activities directly attributed to treatment was USD 225,649 (USD 6676 or **EUR 6300** per kg of PCBs removed) (US Army Corps of Engineers, 1998).

An earlier Superfund PCB case shows much lower costs, more in line with those reported for the Netherlands. At the Outboard Marine Corporation Superfund Site, Waukegan, Illinois, a PCB cleanup (using thermal desorption) took place in 1992. The average PCB content of the untreated soil/sediment was 10,484 mg/kg and 12,755 tons of soil/sediment were treated with an efficiency of 99.98%, so the total amount of PCBs removed was almost 133,700 kg. The total cost of the operation was USD 3,374,000, implying a cost of USD 25 or **EUR 24 per kg** of PCBs removed (EPA, undated a).

Thermal desorption was also applied in another early Superfund case, the Wide Beach Development Superfund Site, Brant, New York (1990-91). Contamination of soil at this site resulted from the spraying of waste oil containing PCBs over the roadways in the community to control dust. In this case the PCBs were also dechlorinated, using alkaline polyethylene glycol. 42,000 tons of soil were treated. The initial average PCB concentration was 24 mg/kg and this was reduced to less than 2 mg/kg, implying a total amount of PCBs removed of at least 924 kg. The total cost was USD 15,908,000, which means a cost of about USD 17,200 or **EUR 16,200 per kg** of PCBs (EPA, undated b).

Cost estimates of PCB remediation in 'open' applications (e.g. in paints and sealants) are few and far between. A number of buildings in Germany have been cleaned up at an average cost of some EUR 100,000 per building (Bruinen de Bruin and Janssen 2012) and the Danish government is spending DKK4.8 mln (EUR 0.7 mln) to help local authorities deal with PCB concentrations in buildings (ENDS 2013)<sup>21</sup>. These cost figures cannot be readily related to specific amounts of PCBs removed. A rough estimate for the German cases can be obtained as follows. If we assume that in the buildings where clean-up has been performed the total initial amount of PCB containing materials was 50 kg, and the PCB content of these materials was on average 5000 mg/kg<sup>22</sup>, we arrive at 250 g PCBs per building, implying an average clean-up cost of **EUR 400,000 per kg** PCBs.

### Inclusion and exclusion of cost categories

Almost all estimates found include the incineration or destruction of the PCBs as hazardous waste. An exception is the cleanup remedy for the entire New Bedford harbor area (EPA, 1998), where disposal was preferred over incineration (the latter was considered to be too expensive). In some cases, the replacement cost of equipment was also included. Whether or not such 'sunk' costs should be included is still a matter of discussion in ECHA (see ECHA 2013, section 3.2). Other estimates (e.g. Neupert, 2004) relate to cases in which the equipment is not replaced, but the PCBs are removed (decontamination) and the equipment can be reused.

Costs of collection, transport, monitoring etc. are sometimes but not not always included in the estimates.

<sup>21</sup> This amount is just for investigations; not for remediation (Janssen, 2015).

<sup>22</sup> In the Danish investigation (Energistyrelsen, 2013) the PCB content in materials exceeded 5000 mg/kg in 9% of the cases. It seems reasonable to assume that such cases are selected for clean-up with priority.

*Box 1 The New Bedford Harbor Superfund Site*

The New Bedford Harbor Superfund Site is located along the northwestern shore of Buzzards Bay in New Bedford, Massachusetts. From the 1940s to 1978, PCB-contaminated wastewater from electronics manufacturing operations was discharged onto the shoreline and into the harbor. Site investigations determined that sediments were contaminated with PCBs and heavy metals. The site was listed on the National Priorities List in September 1983. The Record of Decision (ROD) for a five acre area known as the "Hot Spot area" included dredging of PCB-contaminated sediments followed by incineration. However, due to opposition to incineration, EPA postponed the incineration component of the Hot Spot remedy to explore alternative treatment technologies. In 1996, EPA evaluated four technologies as possible alternatives to incineration - solvent extraction/dechlorination, vitrification, thermal desorption/gas phase chemical reduction, and solidification/stabilization. Reports are available for pilot tests on the first three technologies (EPA, 2000a-c).

*Solvent extraction / dechlorination:* average PCB concentrations in the sediment (five batches) were 2,395 mg/kg. The total cost of treating 18,000 tons of sediment was estimated at USD 12,971,000, implying a cost of USD 301 or **EUR 284 per kg** of PCBs.

*Vitrification:* average PCB concentrations in the sediment were 2,085 mg/kg. The total cost of treating 18,000 tons of sediment was estimated at USD 20,687,000, implying a cost of USD 551 or **EUR 520 per kg** of PCBs.

*Thermal desorption/gas phase chemical reduction:* average PCB concentrations in the sediment were 5,700 mg/kg. The total cost of treating 18,000 tons of sediment was estimated at USD 11,114,000, implying a cost of USD 108 or **EUR 102 per kg** of PCBs.

In 1998, EPA selected a cleanup remedy for the entire harbor area (EPA, 1998). This remedy involved the dredging and containment of approximately 450,000 cubic yards of PCB-contaminated sediment. Threshold levels for the dredging of sediment varied between 1 and 50 ppm of PCBs. The dredged sediments would be placed in four shoreline confined disposal facilities (CDFs). The cost of the remedy was estimated between USD 120 and 130 million. One of the alternatives considered included incineration to destroy the PCB molecules, with an estimated cost of USD 575.9 million. This alternative was not selected since "the extra degree of long term protectiveness provided by having the sediment treated is not considered to be commensurate with the extra cost for that treatment." If we assume (rather arbitrarily, lacking more specific data) an average PCB content in the dredged sediment of 50 ppm, and a density of 1.5 tonnes per cubic yard of sediment, the total amount of PCBs in the dredged sediment can be roughly estimated at almost 35 tonnes. The cost per kg of PCBs of the selected remedy would therefore be some USD 3500 or **EUR 3300 per kg**. The cost of the alternative with incineration, USD 17,000 or **EUR 16,000 per kg** was considered too high.

In 2012, the EPA and the polluter (AVX) reached an agreement by which AVX would pay USD 366 million plus interest for the cleanup of the site. This would be the largest single-site cash settlement in the history of the Superfund program. Under a previous (1992) settlement, AVX had already paid USD 66 million (Department of Justice, 2012).

### *Control or destruction*

Since the PCBs are incinerated or otherwise chemically decomposed, the available cost estimates relate to cases of complete destruction of the substance. As indicated above, the exception is the New Bedford case, where controlled disposal was selected instead of incineration.

### *Factors affecting estimated cost levels*

Economies of scale are important in PCB clean-up, with large items having a 3 to 5 times lower cost than small ones (Friege, 2012). Vermeulen *et al.* (1989) report subsidized costs for capacitors that are 16 times higher than those for transformers, which may also be related to the fact that capacitors are on average smaller and contain a smaller amount of PCBs than transformers.

The concentration of PCBs is another main factor. Near the threshold of 50 ppm the disposal cost of PCBs may well reach the order of magnitude of EUR 10,000 per kg. Related to this is the spatial dispersion, as can be seen from the high cost estimates in cases where groundwater is contaminated with PCBs, and where PCBs were used in 'open' applications (building materials).

In the Sparrevohn Superfund case, the remoteness of the polluted site was a factor affecting the cost level.

### *Evidence on cost considerations in decision making*

The evaluation of the PCB Directive (EC, 2014, chapter 5) concludes that "considerable progress has been made in regards to the inventories and the elimination of PCB. Nevertheless, several factors hindered the achievement of the PCB Directive deadline of 2010 such as the economic crisis, high decontamination costs, and illegal handling and stocking of PCB contaminated equipment." However, it does not mention examples where cost estimates were used in the argumentation to postpone or refrain from PCB removal operations.

As indicated above, from the 500 ppm threshold in combination with the cost estimates in EC (2014), we can derive an implicit estimate in the range of EUR 100 to 1000 per kg that is considered to be acceptable for (mandatory) PCB disposal. From the US, we have some examples where higher amounts (up to EUR 23,000 per kg) were spent, but also one case where an estimated EUR 16,000 per kg was considered too expensive.

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## PFOA

### Introduction

Perfluorooctanoic acid (PFOA) and PFOA-related substances provide special properties, such as high friction resistance, dielectrical properties, resistance to heat and chemical agents, low surface energy, as well as water, grease, oil, and dirt repellency, and are therefore used for various articles, mixtures and applications. Due to its PBT and CMR properties, PFOA and its ammonium salt (APFO) has been identified as substances of very high concern (SVHC) under REACH in July 2013. An Annex XV Restriction Report for PFOA, PFOA salts and PFOA-related substances was presented by the German and Norwegian competent authorities in October 2014 (ECHA, 2014).

### Substitution costs

The Annex XV Restriction Report (ECHA, 2014) states: “Based on the cost and emission estimates the cost-effectiveness of the proposal was assessed with central estimates of <1,649 EUR/kg PFOA and 734 EUR/kg PFOA-related substances emissions reduced. The cost-effectiveness of the proposed restriction in reducing the emissions of PFOA and PFOA-related substances is considered to be proportionate to the risk of PFOA and PFOA related substances taking into account the specific concerns related to these compounds.” (p. 15) This cost-effectiveness is in the same order of magnitude as the cost-effectiveness of existing regulations for other PBT-(like) substances. (p. 150) It is added that “these cost-effectiveness estimates highly depend on the assumptions on substitution costs as well as on emission factors. As the data basis to derive cost as well as emission estimates is very limited, the cost-effectiveness estimates have to be considered as indicative values only.” (p. 174)

The estimated cost-effectiveness range for the substitution of PFOA used in the production of fluoropolymers (the only remaining major application of PFOA in the EU) by other polymerisation processing aid compounds<sup>23</sup> was **EUR 0 to EUR 6,561 per kg** PFOA emission avoided (with the above mentioned central estimate of **EUR 1,649**), and **EUR 0 to EUR 2,493 per kg** PFOA substituted (with a central estimate of **EUR 627 per kg** (ECHA, 2014, Table F.2-7). For PFOA-related substances, where the alternatives mainly consist of short-chain fluorotelomers<sup>24</sup>, the substitution cost range was estimated between **EUR 4 and EUR 3,533 per kg** avoided emissions (central estimate **EUR 734**) and between **EUR 2 and EUR 53 per kg** of PFOA-related substances substituted (central estimate **EUR 14**).

A number of other substitution cost estimates relate to both PFOA and PFOS. These are discussed in the next chapter on PFOS.

### *Inclusion and exclusion of cost categories*

The mentioned substitution cost estimates in ECHA (2014) are based upon assumptions (mainly based on information provided by industry) regarding the expected increases in production costs, in the price of the substitutes and the additional amounts of these substitutes that would be needed to achieve an equivalent technical performance.

<sup>23</sup> Based on the relative price increase of PTFE (fluoropolymer with main market share) manufactured without PFOA and PTFE price

<sup>24</sup> Based on relative price increase of short-chain alternatives, the price range of PFOA-related substances and the additional volumes to be used to achieve an equivalent technical performance.

### *Functionality and performance issues*

For some applications, the alternatives for PFOA-related substances may not be perfect substitutes. For example, for technical textiles the change to alternatives could result in a loss in product quality in terms of oil and dirt repellency, which could be decisive for the utility of the respective product (ECHA, 2014, p. 169).

### *Factors affecting estimated cost levels*

The wide ranges in the cost estimates in ECHA (2014) are mainly due to the large uncertainties regarding both the amounts of PFOA and PFOA-related substances used and emitted as well as the cost of substitution in the different applications.

### *Evidence on cost considerations in decision making*

As the restriction proposal has not yet been decided upon, no conclusions can as yet be drawn concerning the role of cost considerations in decisions on PFOA restrictions in the EU.

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## PFOS

### Introduction

In the past, perfluorooctane sulfonate (PFOS) had many applications, including in products such as textiles, leather, carpets, paint, paper, cardboard and firefighting foam. In 2009, PFOS was added to Annex B of the Stockholm Convention, implying that Parties must take measures to restrict the production and use of PFOS, but leaving a number of applicable acceptable purposes and specific exemptions. In the EU, the production, placing on the market and use of PFOS is prohibited (Regulation 850/2004/EC as amended by Regulation 757/2010/EU). Exemptions are allowed for the use of PFOS as:

- wetting agents for use in controlled electroplating systems (until 26 August 2015);
- photoresists or anti reflective coatings for photolithography processes;
- photographic coatings applied to films, papers, or printing plates;
- mist suppressants for non-decorative hard chromium (VI) plating in closed loop systems;
- hydraulic fluids for aviation.

### Substitution costs

Estimates of substitution costs for PFOS show a high variability, both within and between different areas of application (see table below<sup>2</sup>). For surface protection, metal plating (if drop-in substitutes are available) and firefighting foam, alternatives appeared to be available at low additional cost; in some cases close to zero or even negative. Relatively high costs are found in the applications that are (in the EU) exempted from the ban on PFOS use: metal plating (if the only feasible alternative is ventilation or other mechanical solutions); photographic industry; photolithography and semiconductors. Information on the cost of alternatives for PFOS in hydraulic fluids in aircraft is lacking.

*Estimated substitution costs for PFOS (EUR per kg)*

Application	Low estimate	High estimate	Source
Coatings and paints	<0		UNEP (2012)
Firefighting foams	0	201	RPA (2004); UNEP (2007)
Metal plating (drop-in)	-100	1,000	RPA (2004); HELCOM (2013)
Metal plating (mechanical)	40	400,000	UNEP (2007); HELCOM (2013)
Photolithography and semiconductors	500,000	8,000,000	HELCOM (2013)
Photographic materials	1,500	28,000*	RPA (2004); UNEP (2007); HELCOM (2013)

\* The EUR 7 mln per kg mentioned in HELCOM (2013) is apparently an error; it should be EUR 7000 per kg (and therefore lies within the range mentioned here).



### *Inclusion and exclusion of cost categories*

The estimated substitution costs for PFOS generally include both the capital cost of the necessary new equipment and the operational costs. In several cases the cost of the alternative substance is lower than the cost of PFOS, implying savings on operational costs.

### *Functionality and performance issues*

The equivalence of the substitute varies by area of application. For example, the RPA (2004) report, which underlies many of the other substitution related reports, mentions some disadvantages of substitutes for PFOS in the use area of metal plating, whereas for PFOS in firefighting foam the available substitutes are considered to be equivalent.

### *Factors affecting estimated cost levels*

High per kg substitution costs for PFOS are mainly found in those application areas where the total amount of PFOS used is relatively small. However, in some cases the estimated costs are not related to the total use, but to the PFOS emission levels. For example, in the UNEP (2007) report the cost of substituting the remaining use of PFOS in photo imaging is estimated at USD 50 million, whereas only 2 kg is released into the environment, implying a cost of USD 25 mln (EUR 22 mln) per kg. If the total cost is related to the total amount of PFOS still used (10 tonnes) the amount per kg (USD 2500) would be much lower.

### *Costs of disposal, emission control and remediation*

This type of cost data mainly relates to PFOS in firefighting foam and to the removal of PFOS from waste water and drinking water supplies.

Replacing firefighting systems that contain PFOS is generally considered to be relatively cost effective. Destruction costs were estimated at about EUR 1000 per tonne of foam and replacement costs about EUR 4000 to 5000 per tonne (RPA, 2004, p. 139; LRTAP, 2006). The PFOS content is about 10 kg per tonne of foam (calculated on the basis of RPA, 2004, p. iii), which means the destruction and replacement cost for this application would together be about **EUR 500 to 600 per kg PFOS**. Hellema (2015) provided us with an estimate of the replacement costs of 6000 litres of PFOS containing foam: around EUR 60,000. Assuming a PFOS content of around 15 grammes per litre of foam (based on the data from the Schiphol case; see Textbox 3), this would mean some **EUR 700 per kg PFOS**. However, in specific cases the cost can be much higher (see Textbox 2).

Several cases of PFOS polluted sites exist where firefighting foam with PFOS has been used. Estimated clean-up costs of these sites (e.g. Buncefield, Jersey, Schiphol; see Textbox 3) are generally in the order of magnitude of EUR 1 mln or more per case (Nicholas and Whitfield, 2013; Klein, 2013; Gedeputeerde Staten, 2011). More information on the total amount of PFOS that has entered the environment (and is or will be removed) would be needed to arrive at an average per kg estimate for each of these cases.

*Box 2 The Barendrecht case*

According to the POP regulation (850/2004, implementing the Stockholm Convention) the use of PFOS in firefighting systems is prohibited since 28 June 2011. Maximum residual concentrations of 10 mg/kg are allowed.<sup>25</sup>

In anticipation of this deadline, Dutch railway infrastructure company ProRail carried out a project to replace PFOS in the firefighting system in the Barendrecht railway tunnel. The initial PFOS concentration in the installation was estimated between 700 and 7500 mg/kg. Flushing out did not result in a reduction below the legal threshold. According to ProRail, this was due to the fact that the substance is sticky and that materials can take up PFOS that is released again in a later stage. The PFOS content was reduced with a factor 100 to 700, which implies that concentrations of up to 75 mg/kg could still be present.<sup>26</sup> The cost of the operation was about EUR 700,000. A second round of flushing out would again cost the same amount, but there might still remain places where the standard would not be achieved. Full compliance, according to ProRail, would only be feasible if the main pipe of the firefighting installation would be replaced. The cost of that operation was estimated at EUR 1 to 2.5 million.

According to calculations by Ewald van Dorst (Marak) the remaining amount of PFOS in the system after the flushing out was almost 1.8 kg.<sup>27</sup> This would imply that the cost effectiveness of a complete PFOS removal by replacing the main pipe would be **between EUR 600,000 and 1.4 million per kg PFOS**. Should the legal limit be achievable by a second flushing out operation, then the cost effectiveness would be almost **EUR 400,000 per kg PFOS** (EUR 0.7 mln to remove 1.8 kg).

The cost of the initial operation (which did not lead to complete PFOS removal) can be estimated as follows. According to the documents from the waste treatment company (Indaver), a total amount of 26,460 kg firefighting foam was removed. Given the above-mentioned concentrations (700 to 7500 mg/kg) this means an amount of PFOS between almost 20 and 200 kg. Since the cost of the operation was EUR 700,000, the cost effectiveness was **between EUR 3,500 and 35,000 per kg**.

To our knowledge, ProRail has until now not been obliged to perform additional PFOS cleaning operations. However, there is also no official statement from the authorities that ProRail's argument (that the cost of such additional operations would be disproportionately high) is accepted and that exceedance of the legal limit would be tolerated. Therefore, we can conclude in this case to a minimum implicit social willingness to pay of EUR 3,500 per kg, but no maximum can be determined.

<sup>25</sup> This is the maximum concentration in substances and preparations. For waste containing PFOS a limit of 50 mg/kg is applied (above this limit the waste should be treated in such a way as to ensure that the PFOS content is destroyed or irreversibly transformed so that the remaining waste and releases do not exhibit the characteristics of persistent organic pollutants). For the analysis done here the question which of the two limits applies is not relevant.

<sup>26</sup> According to calculations by Peter Hellema of the Environment and Transport Inspectorate, the average residual concentration was 35 mg/kg.

<sup>27</sup> Peter Hellema of the Environment and Transport Inspectorate estimated the maximum remaining amount at 1.675 kg.

*Box 3 The Schiphol case*

On July 12, 2008, a sprinkler installation in a hangar at Schiphol airport came accidentally into operation, releasing among others firefighting foam with PFOS. According to Gedeputeerde Staten (2011) 10,000 litres of foam were released, containing 143 kg PFOS. Some 100 mln litres of polluted surface water were temporarily stored in five basins. However, four of these appeared to leak, resulting in groundwater and surface water pollution in a wide area.

In order to avoid further diffusion of the pollution, a bentonite screen (9 meters deep) was put around an area of 16 hectares. Water within the enclosure is drained, purified and discharged. During the first two years of this containment project (2012-2014) the costs were EUR 2 million (Heijmans 2014). The water board has made a reservation of EUR 2.6 million to deal with the issue (Rijnland, 2011). In addition, EUR 3.2 million is budgeted by the province of Noord-Holland to clean up a part of the contaminated area that is needed for the construction of a new road. The full sanitation operation is estimated to cost between EUR 30 and 40 million (Gedeputeerde Staten, 2011).

The more than EUR 5 million that has been spent or budgeted until now to immobilize and remove (less than) 143 kg of PFOS implies an implicit social willingness to pay of at least **EUR 35,000 per kg** PFOS. If it will be decided to complete the full sanitation this could increase to between **EUR 200,000 and 300,000 per kg**.

Other 'end-of-pipe' cases are related to the removal of PFOS from wastewater or from surface water used for drinking water supplies. Here we find estimates ranging **from several thousands up to several millions of euros per kg** of PFOS removed (see Helcom, 2013; see also Text box 4).

The cost of incinerating contaminated aqueous waste originating from developer processes in the semiconductor industry is estimated in RPA (2004) at EUR 3.9 million per year across the EU to destroy 195 kg of PFOS, i.e. **EUR 20,000 per kg**.

*Inclusion and exclusion of cost categories*

The cost estimates for disposal, emission control and remediation of PFOS usually only cover the direct costs related to the operations (including, where applicable, the cost of decomposing the substance by means of incineration). Some estimates also include indirect costs; for example, the highest cost estimate in the Barendrecht case accounted for the fact that replacing the main pipe of the firefighting system would require closure of the tunnel for some time, resulting in economic losses.

*Box 4 The Arnsberg case*

The public water supply company in Arnsberg (Germany) operates an active coal treatment plant to remove PFOS and PFOA from surface water that it receives from the Möhne reservoir. The water is polluted as a result of the use of contaminated sewage sludge as fertilizer in agriculture and forestry. In 2006 and 2007, the water company incurred annual costs of some EUR 300,000 for this water purification (Arnsberg, 2007). In the wider Möhre and Ruhr area, several other measures aiming at sanitation and water purification due to PFOS/PFOA pollution have been taken. In 2007, an amount of EUR 60 mln was foreseen for short term investments in these measures (Nordrhein-Westfalen, 2007).

In the Möhne reservoir, PFOS concentrations of 135 to 405 ng/l were reported, and PFOA concentrations of 11 to 7,070 ng/l (Umweltbundesamt, 2009). Stadtwerke Arnsberg provides 10.5 mln litres of drinking water per day (Arnsberg, 2015), or 3.8 mln m<sup>3</sup> per year. The share of water from the Möhne reservoir in this supply is unknown. To arrive at a conservative estimate, we assume that the raw water for all 3.8 mln m<sup>3</sup> contained the maximum concentration of PFOS and PFOA (i.e. together almost 7.5 mg/m<sup>3</sup>) and is purified by the active coal treatment. This would mean that per year some 28.5 kg of PFOS and PFOA is removed, which implies an implicit social willingness to pay of at least **EUR 10,000 per kg**. Given the conservative assumptions, the amount per kg could easily be one or two orders of magnitude higher.

In 2009, the Court in Arnsberg issued a verdict (VG Arnsberg, 2009) in which the sanitation costs of agricultural land that was polluted by some 400 kg PFOS and PFOA were estimated at EUR 2.7 mln, which comes down to **EUR 6750 per kg**.

*Control or destruction*

PFOS that is removed from firefighting systems is usually incinerated in chemical waste incineration plants. The same is true for PFOS (and PFOA) that is removed from waste water or surface water. In some soil pollution cases, however (for example Schiphol) the focus is on preventing the wider dispersion of the substance by creating barriers. Only in those parts where new developments are planned (such as, in this case, road construction) full remediation takes place.

*Factors affecting estimated cost levels*

PFOS is a very mobile substance. Therefore, the cost of remediation in case of a PFOS spill will increase rapidly with time, as a larger area becomes polluted.

*Evidence on cost considerations in decision making*

Restrictions on the marketing and use of PFOS were in the EU initially introduced by Directive 2006/122/EC, and later on transferred to Annex XVII of the REACH Regulation (by Regulation 552/2009) and then to the POP Regulation (by Regulation 757/2010). A number of PFOS uses are exempted from the general ban on PFOS use: wetting agents for use in controlled electroplating systems (until 26 August 2015); photoresists or anti reflective coatings for photolithography processes; photographic coatings applied to films, papers, or printing plates; mist suppressants for non-decorative hard chromium (VI) plating in closed loop systems; and hydraulic fluids for aviation. As noted above, these applications coincide with the uses where relatively high substitution costs were reported. Nonetheless, the EU legislation does not

explicitly mention the high cost to motivate the derogations. It does say, however, that the exempted uses of PFOS “will be phased out as soon as the use of safer alternatives is technically and economically feasible”.

As the Barendrecht, Schiphol and Arnsberg cases show, costly measures are actually taken to remove PFOS from existing stocks and from the environment. When estimating the costs of such measures on a per kg basis, the outcomes tend to be in the same orders of magnitude as the costs of substitution for the exempted PFOS applications. Such explicit per kg estimates are, however, usually not made in the decision making on sanitation/remediation cases. In any case, the discretionary competence for the authorities is limited in such cases, since standards (e.g. for drinking water quality) have to be met anyway.

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